Breeding and persistence of woodland birds in restoration plantings

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Cover photo: A male flame robin (*Petroica phoenicea*) captured in a restoration planting in the South-west Slopes bioregion, NSW. Photo: Donna Belder.

DECLARATION

This thesis is my own work, except where otherwise acknowledged.

Donna Belder 22 April 2020

PREFACE

Structure of the thesis

With the exception of the context statement (introduction) and synthesis (conclusion), this thesis is presented as a series of connected manuscripts on a related theme. Each manuscript is intended as a separate scientific paper – for this reason, there is some unavoidable repetition in the content, particularly in the background material and methods. Manuscripts were co-authored with members of my supervisory panel and other academics, and for this reason are written in the plural first person ("we" instead of "I"). At the time of submission, manuscripts were either published in peer-reviewed journals, under revision, or in preparation for submission to a peer-reviewed journal. The content of each chapter is the same as the manuscripts that have been published/reviewed/prepared for submission, but formatting and numbered items have been modified for consistency throughout the thesis. Due to journal copyediting, there may also be some minor differences in the versions that appear in this thesis compared with the final, published versions.

The structure and format of this thesis comply with the Australian National University standards for a "Thesis by Compilation". The thesis begins with a context statement that serves as a general introduction to the thesis, outlining the relevance of the research and detailing the context of the work in the scope of the broader research area. The context statement is not intended as an extensive literature review, as each chapter introduction reviews the literature relevant to that chapter. The first chapter of the thesis is a published, comprehensive literature review of the general topic on which this thesis is based.

I performed the majority of work for the papers presented in this thesis. I was solely responsible for managing the research project, and conducted the bulk of all tasks including:

designing and conceptualising the study, researching the topic, selecting field sites, liaising with landowners, obtaining relevant ethics approvals and research permits, applying for funding, managing budgets, organising and conducting fieldwork, sourcing and supervising volunteers, data-entry, data analysis, writing manuscripts, presenting the research at workshops, seminars and conferences, and writing the thesis. My supervisory panel and other co-authors assisted with various aspects of the project. My supervisors (David Lindenmayer, Jennifer Pierson, Karen Ikin, Mason Crane, and Ayesha Tulloch) helped design and conceptualise the study, advised on analysis, and contributed to manuscript revisions at various stages of the project. I also sought advice from statisticians and other academics within the Fenner School, including Wade Blanchard, Martin Westgate, and Jeff Wood. David Lindenmayer and colleagues, including Mason Crane, are responsible for developing and maintaining the long-term South-west Slopes Restoration Study that forms the basis for this project. Other assistance is detailed in the acknowledgements section of each paper.

Author contributions

The list of citations and author contributions is described below.

Chapter 1

Belder D. J., Pierson J. C., Ikin K., and Lindenmayer D. B. (2018). Beyond pattern to process: current themes and future directions for the conservation of woodland birds through restoration plantings. *Wildlife Research* **45** , 473–489.

DJB wrote the first draft of the manuscript and oversaw revisions with the input of the other co-authors. DBL, JCP, and KI contributed to manuscript revisions and advised on content and structure.

Chapter 2

Belder, D. J., Pierson, J. C., Ikin, K., Blanchard, W., Westgate, M. J., Crane, M., and Lindenmayer, D. B. (2019). Is bigger always better? Influence of patch attributes on breeding activity of birds in box-gum grassy woodland restoration plantings. *Biological Conservation* **236**, 134–152.

DJB conducted fieldwork, analysed data, wrote the initial draft of the manuscript, and revised the manuscript according to peer review comments. JCP, KI, and DBL contributed to manuscript revisions and advised on content, framing, and structure. WB and MJW provided advice on statistical analysis and assisted with R coding. MJW also responded to a peer review comment requesting further information on multivariate latent variable models. DBL and MC helped design the study and select study sites, and managed the historical bird survey database that was used to select study sites.

Select findings from the research detailed in Chapter 2 were presented at the *Annual Conference for the Ecological Society of Australia 2018*:

Belder, D.J. (2018). Is bigger always better? Influence of patch size on breeding activity of woodland birds in restoration plantings. Paper presented at the *Annual Conference for the Ecological Society of Australia 2018*, Brisbane, Australia, 25-29 November 2018.

Chapter 3

Belder, D. J., Pierson, J. C., Ikin, K., and Lindenmayer, D. B. (2020). Revegetation and reproduction: Do restoration plantings in agricultural landscapes support breeding populations of woodland birds? *Oecologia* **192**, 865–878: Highlighted Student Research.

DJB conducted fieldwork, analysed data, wrote the initial draft of the manuscript, and prepared the manuscript for submission. JCP, KI, and DBL contributed to manuscript revisions, and advised on content, framing, and structure. DBL helped design the study and select study sites, and managed the historical bird survey database that was used to select study sites.

Preliminary findings arising from the research described in Chapter 3 were presented at the *28th International Congress of Conservation Biology*:

Belder, D. J. (2017). Breeding success of woodland birds in Australian box-gum grassy woodland restoration plantings. Paper presented at the *28th International Congress of Conservation Biology*, Cartagena, Colombia, 23-27 July 2017.

Chapter 4

Belder, D. J., Pierson, J. C., Rudder, A. C., and Lindenmayer, D. B. (2020). Ongoing declines of woodland birds: are restoration plantings making a difference? *Ecological Applications* (under review).

DJB conducted fieldwork, analysed data, wrote the initial draft of the manuscript, and prepared the manuscript for submission. JCP and DBL provided advice on content, framing, and structure and contributed to manuscript revisions. JCP also assisted with interpreting mark-recapture analysis and participated in one of the bird banding trips. ACR assisted with fieldwork, data-entry and data analysis.

Appendix A

Belder, D. J. (2018). Likely depredation of a Superb Fairy-wren *Malurus cyaneus* nest by an uncommon predator, the White-browed Babbler *Pomatostomus superciliosus*. *Australian Field Ornithology* **35**, 146–148.

DJB is the sole author and conducted all necessary work to produce the manuscript.

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THESIS ABSTRACT

Habitat loss and fragmentation are major influences on the distribution and composition of ecological communities, and are also linked to a loss of agricultural productivity due to ecosystem degradation. As such, there is increasing impetus for the implementation of revegetation in fragmented agricultural landscapes worldwide. Restoration plantings are often intended to provide habitat for threatened fauna and restore ecosystem health. In Australia, loss of temperate woodland habitat has caused the decline of many woodland bird species. There have been numerous studies investigating how bird community richness and abundance is influenced by restoration plantings and other woodland patches in fragmented agricultural landscapes, but this "pattern-focused" research does not provide evidence that restoration plantings can support resident populations of woodland birds. This approach limits the ability of land managers and scientists to assess whether woodland patches provide suitable habitat for woodland birds, and therefore to effectively assess whether restoration plantings are fulfilling their fundamental purpose as a conservation strategy.

In this thesis, I address this knowledge gap through an empirical research project undertaken in the South-west Slopes bioregion of New South Wales, Australia. The series of papers presented in this thesis examines the research questions in increasing detail, beginning with a comprehensive literature review that places the remaining chapters in context of the broad knowledge gap (Chapter 1), then recording evidence of breeding activity (Chapter 2), monitoring nest success and daily nest survival (Chapter 3), and examining the home ranges and annual survival of individually tagged woodland birds (Chapter 4). These papers collectively represent a detailed study of the responses of woodland birds to box-gum grassy woodland restoration plantings in a fragmented agricultural landscape. I offer management

implications of each chapter's key findings in the relevant discussion sections, and conclude the thesis with a synthesis of applications and directions for future research on the topic.

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CONTEXT STATEMENT

"Birds were flying from continent to continent long before we were. They reached the coldest place on Earth, Antarctica, long before we did. They can survive in the hottest of deserts. Some can remain on the wing for years at a time. They can girdle the globe. Now, we have taken over the earth and the sea and the sky, but with skill and care and knowledge, we can ensure that there is still a place on Earth for birds in all their beauty and variety – if we want to – and surely, we should."

– Sir David Attenborough

Background

Global biodiversity and habitat loss

Habitat loss is one of the greatest threats to the world's biodiversity (Tscharntke *et al.* 2012; Maxwell *et al.* 2016; Tilman *et al.* 2017). Habitat loss and fragmentation are increasing worldwide, particularly in highly productive agricultural regions, and anthropogenic climate change is widely predicted to exacerbate existing patterns of biodiversity decline and ecosystem degradation (Bennett *et al.* 2015a; Hoegh-Guldberg *et al.* 2018; Malhi *et al.* 2020). Scientists have called for intensified conservation efforts to halt a so-called "sixth mass extinction" caused by human impacts on the global environment (Wake and Vredenburg 2008; Barnosky *et al.* 2011; Ceballos *et al.* 2015).

Habitat loss in Australia

Australia is a global hotspot for biodiversity declines due to a range of factors, including feral predators (Woinarski *et al.* 2015; Allek *et al.* 2018), inappropriate land management (Cresswell and Murphy 2017), and habitat loss (Kerle *et al.* 2014; Reside *et al.* 2017; Ward *et al.* 2019). While much of the land clearing in Australia's southern temperate regions occurred within the first century of European colonisation (Bradshaw 2012), the rate of deforestation in Australia remains one of the highest in the developed world (Evans 2016). This is largely

attributed to authorities such as the respective state governments of Queensland (Reside *et al.* 2017) and New South Wales (Bartel and Graham 2016) weakening protections for native vegetation and facilitating the acceleration of broad-scale land clearing. Australia's federal environmental legislation is also ineffective in preventing habitat loss for threatened species (Ward *et al.* 2019).

Temperate woodlands

Temperate woodlands once covered a significant portion of southern Australia. Their historic extent included the central-west of NSW to the Great Dividing Range, a large part of Victoria, the Mount Lofty Ranges of South Australia, and the Great Western Woodlands in south-west Western Australia (Figure 1). These areas have all been intensively cleared, mainly for livestock grazing. In some regions, over 85% of temperate woodlands have been cleared (Lindenmayer *et al.* 2010a), and nearly two decades ago, it was estimated that only 0.1% of box-gum grassy woodlands remained in intact condition (Thiele and Prober 2000). In addition to ongoing land clearing, other key threatening processes in temperate woodlands include nutrification, grazing, weed invasion, climate change, and competitive exclusion of small woodland birds by the noisy miner (*Manorina melanocephala*), a hyperaggressive native honeyeater (Lindenmayer *et al.* 2010a).

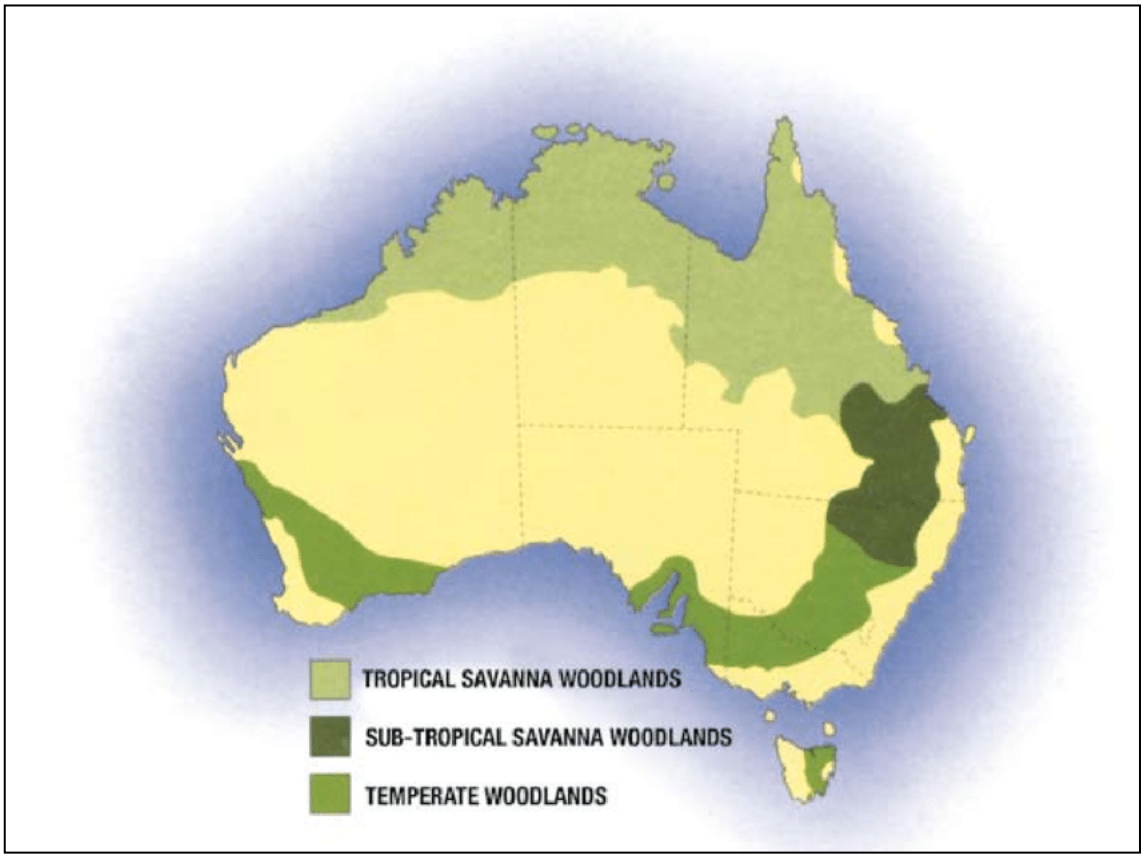

Figure 1 The historic distribution of temperate, sub-tropical and tropical woodlands in Australia. Image from Watson et al. (2008).

Box-gum grassy woodlands

Box-gum grassy woodland remnants are scattered across the largely cleared agricultural regions of south-eastern Australia, ranging in size from national parks, state forests and travelling stock reserves, through to narrow roadside verges and scattered paddock trees (Gibbons and Boak 2002; Harwood and Mac Nally 2005; Lentini *et al.* 2011). Up to one quarter of total remnant cover is comprised of small, isolated patches of less than 5 ha on private land (Duncan and Dorrough 2009). These privately-managed remnant patches are often in poor to severely degraded condition (Eldridge 2003; Duncan and Dorrough 2009; Lindenmayer *et al.* 2010a). Several kinds of box-gum grassy woodlands are listed as threatened under the EPBC Act, including White Box-Yellow Box-Blakely's Red Gum Grassy Woodland and Derived Native Grassland, Poplar Box Grassy Woodland on Alluvial Plains, Grey Box Grassy Woodlands and Derived Native Grasslands of South-Eastern

Australia, and Peppermint Box Grassy Woodland of South Australia (Department of Agriculture, Water and the Environment 2020).

Effects of habitat loss and fragmentation on birds

Many bird species associated with Australian temperate woodlands are suffering population declines due to loss of habitat and isolation of woodland remnants (Ford 2011; Rayner *et al.* 2014a; Lindenmayer *et al.* 2018a). Woodland-dependent species feature heavily on threatened species lists in Australia at both state and federal levels (Olsen *et al.* 2005; NSW Office of Environment and Heritage 2018; Department of Environment and Energy 2019). In this thesis, I consider species of conservation concern to be those that are listed as Vulnerable, Endangered, or Critically-endangered at either a federal or NSW state level, as well as species for which a significant decline has been identified based on differences in reporting rates between the first and second Atlas of Australian Birds (Barrett *et al.* 2003 vs. Barrett *et al.* 2007).

Habitat loss and degradation are considered the underlying causes of declines for woodland bird species of conservation concern, but there are multiple aspects of these processes that may affect survival and recruitment of birds in fragmented agricultural landscapes:

Area-sensitivity and edge-avoidance

There is substantial evidence that area-sensitivity has led to the decline of many bird species in fragmented landscapes. Area-sensitivity is the absence of a species from smaller habitat patches in a fragmented landscape (Zanette *et al.* 2000), and can be linked to various factors. For example, area-sensitivity in the eastern yellow robin (*Eopsaltria australis*) is attributed to greater long-term stress and reduced reproductive success associated with fewer available

invertebrate resources in smaller woodland fragments (Zanette *et al.* 2000; Maron *et al.* 2012). Patch geometry and the ratio of edge to interior habitat, which are interrelated with patch area, also have highly influential effects on some bird species' abundance and distribution patterns (Davis 2004; Harwood and Mac Nally 2005).

While area-sensitivity is often linked to edge-avoidance, not all declining birds are negatively affected by edges. For example, the endangered hooded robin (*Melanodryas cucullata*) is a woodland-dependent species that is positively associated with edges of open vegetation dominated by an ungrazed or lightly grazed grassy ground cover (Priday 2010). However, it is important to note that birds may respond differently to natural and induced edges (Luck *et al.* 1999).

Loss of connectivity

Functional connectivity in fragmented landscapes is often significantly reduced, impeding the movement of birds between habitat patches (Bélisle 2005; Doerr *et al.* 2011). Some bird species in isolated patches are reluctant to cross habitat gaps of only around 60 m (Brooker *et al.* 1999; Radford *et al.* 2005). This has substantial consequences for gene flow, dispersal, and recruitment of bird species in fragmented agricultural environments (Ford 2011; Sunnucks 2011). For example, isolation-sensitivity due to impaired dispersal is thought to be one of the primary reasons for the decline of the brown treecreeper (*Climacteris picumnus*) in fragmented temperate woodlands (Cooper and Walters 2002; Cooper *et al.* 2002; Doerr *et al.* 2011). Social birds such as the white-browed babbler (*Pomatostomus superciliosus*) are also negatively affected by habitat loss and fragmentation; occupying smaller and more isolated habitat patches is likely to decrease social interactions and hence productivity (Cale 2003a; Cale 2003b).

Reduced survival and recruitment

Few studies have directly tracked the survival of individual birds in temperate woodlands. A study of the red-capped robin (*Petroica goodenovii*) in woodland remnants by Major and Gowing (2001) found no significant difference in the survival rates of adult birds in roadside remnants vs. large (intact) remnants. However, the authors acknowledged that their model was weak due to the low numbers of individuals available for sampling. Major *et al.* (1999) had previously found that the average density of males of the red-capped robin was greater in large remnants than in roadside remnants, but were unable to attribute the differences to site linearity due to confounding between patch shape and size. Luck (2003) studied the reproductive success and survival of the rufous treecreeper (*Climacteris rufa*) in fragmented and intact landscapes. Greater nest success, higher juvenile survival rates and a higher average biomass of prey fed to nestlings contributed to significantly higher annual productivity in intact landscapes, despite there being no differences in nest predation or fledging rates between the two landscape categories. The brown treecreeper is similarly affected by habitat fragmentation, with survival of females negatively associated with poorly-connected remnant patches (Cooper *et al.* 2002).

Predicting declines

Barrett *et al.* (2007) identified a long list of declining woodland-dependent bird species, many of which were previously reported as declining by Reid (1999). Since these publications, the labelling of a species as being of "conservation concern" has varied widely. Species commonly regarded as declining and of conservation concern in temperate woodlands include the scarlet robin (*Petroica boodang*), flame robin (*P. phoenicea*), hooded robin, crested shriketit (*Falcunculus frontatus*), restless flycatcher (*Myiagra inquieta*), diamond firetail (*Stagonopleura guttata*), weebill (*Smicrornis brevirostris*), and various thornbills (*Acanthiza*

spp) (Barrett *et al.* 2007; Paton and O'Connor 2009; Ford 2011; Rayner *et al.* 2014a). Some species of conservation concern have been classified as stable, including red-capped robin, jacky winter (*Microeca fascinans*), grey-crowned babbler (*Pomatostomus temporalis*), southern whiteface (*Aphelocephala leucopsis*), black-chinned honeyeater (*Melithreptus gularis*), brown treecreeper, speckled warbler (*Pyrrholaemus sagittatus*), and rufous whistler (*Pachycephala rufiventris*) (Barrett *et al.* 2007). Many so-called "stable" species are nonetheless uncommon to rare in remnant woodlands, having suffered historic declines since European settlement (Barrett *et al.* 1994; Ford *et al.* 2001). Rayner *et al.* (2014b) argue that to properly address woodland bird conservation, a more robust empirical approach to measuring species decline is needed.

Previous research has established that particular behavioural traits or habitat preferences may place some birds at greater risk of decline than others. Insectivorous species decline before other foraging 'guilds', perhaps due to their selective foraging habits (Lunney *et al.* 1997; Barrett *et al.* 2007; Razeng and Watson 2012). Ground-foraging birds are also susceptible to declines in temperate woodlands (Antos and Bennett 2006; Barrett *et al.* 2007), but this is not necessarily linked to specific or restricted habitat preferences (Antos *et al.* 2008). The latter two generalised findings are not shared by all regional studies of bird reporting rates, and population trends may not be regionally consistent (Lindenmayer and Cunningham 2011). Many of the above-mentioned species of conservation concern are sensitive to disturbance, and their persistence depends on high-quality, intact woodland (Barrett *et al.* 1994; Watson *et al.* 2001; Maron 2007). A tolerance to fragmentation and disturbance is rare in declining species (Reid 1999). The yellow-rumped thornbill (*Acanthiza chrysorrhoa*) is one declining species that appears to persist in fragmented landscapes (Barrett *et al.* 1994; Watson *et al.* 2001). The risk of decline by bird species in fragmented habitats may be lower if they are able

to modify their foraging behaviour as available substrates and prey availability change (Miller and Cale 2000). This may explain why many of the species identified as 'increasers' in fragmented and degraded ecosystems, such as the willie wagtail (*Rhipidura leucophrys*), are generalist foragers that are able to exploit a wide range of habitat types and structures (Maron and Lill 2005).

Climate change may introduce a new suite of problems for woodland birds, causing further changes to habitat quality and resources (Mac Nally *et al.* 2009; Cunningham *et al.* 2013; Bennett *et al.* 2015a). Rising temperatures are expected to exacerbate droughts in south-eastern Australia (Cai *et al.* 2009; Feng *et al.* 2019), which may push populations to the brink of extinction. During the Millennium Drought, 42-62% of woodland bird species surveyed by Bennett *et al.* (2014) declined. Only 21-29% bird species increased again when the drought broke, while more than half did not recover. Furthermore, 21-27% of species continued to decline post-drought. In addition, the species that declined the most during the drought were not necessarily among those more likely to recover well when conditions improved. Rising temperatures and extreme weather events may also decrease avian survival and reproductive success (Bolger *et al.* 2005; Robinson *et al.* 2007; Skagen and Yackel Adams 2012; Cunningham *et al.* 2013; Gullett *et al.* 2015; Moreno *et al.* 2015; Öberg *et al.* 2015; Conrey *et al.* 2016; Martin *et al.* 2017).

Extinction debts

It is possible that many woodland bird species that currently occur in box-gum grassy woodlands are facing extinction debts (Ford *et al.* 2009). An extinction debt is the continued decline of a population or species once a critical threshold of a key threatened process is reached, regardless of that process ceasing, or indeed, reversing (Kuussaari *et al.* 2009). This

is particularly relevant for many species of Australian birds, which are relatively long-lived (20+ years is common in many species weighing 5-50 g) (Australian Bird and Bat Banding Scheme 2016). This is because long generation times may mask recurrent failures of resident populations to breed successfully, or high juvenile mortality rates. A scarcity of well-connected habitat is also likely to reduce recruitment into the population present in the landscape, since dispersal is impeded in fragmented landscapes (Cooper and Walters 2002; Lehnen and Rodewald 2009; Doerr *et al.* 2011; Winiarski *et al.* 2017). Juveniles face precarious journeys across the agricultural matrix to reach new habitat patches, or risk being outcompeted by the resident population in their natal patch (Fahrig 2007).

Restoration attempts

To combat habitat loss and fragmentation in box-gum grassy woodlands and restore ecosystem function and habitat for fauna such as woodland birds, there is an increasing drive to implement restoration plantings, particularly on private land (Smith 2008). Restoration plantings in highly fragmented agricultural landscapes aim to increase habitat quality and connectivity for native wildlife, including threatened and declining woodland birds. Previous studies have found that some restoration plantings play a significant role in increasing bird species richness and abundance on farms (e.g. Lindenmayer *et al.* 2010b). However, we still know very little about the survival and persistence of bird populations in agricultural landscapes undergoing restoration. This may be one reason that current conservation and restoration efforts often do not adequately reflect the habitat requirements of woodland bird species (Montague-Drake *et al.* 2009; Allan 2016).

Temperate woodland restoration plantings typically comprise small blocks of planted native trees and shrubs in paddocks, or linear strips of vegetation between paddocks (also known as

windbreaks) (Lindenmayer *et al.* 2013). Species planted may or may not be indigenous to the locality (Loyn *et al.* 2009; Atyeo and Thackway 2009). Over the past few decades, multiple restoration programs initiated by both government and private sectors have cost billions of dollars (Lindenmayer *et al.* 2013; Campbell *et al.* 2017; Reside *et al.* 2017).

Challenges of restoration

Establishing restoration plantings can be challenging. Various social and financial factors currently impede the widespread implementation of restoration plantings in productive agricultural landscapes (Manning *et al.* 2006; Smith 2008). Acquiring land, obtaining resources (plants, personnel, fencing materials), and ongoing maintenance requirements may variously contribute to either a low uptake or unsuccessful implementation of restoration plantings as a conservation strategy. Fencing, at least during the early stages of growth, is critical to ensure plants are protected from grazing and survive beyond the seedling stage (although it is better to maintain a fence indefinitely) (Lindenmayer *et al.* 2013). However, fencing is also one of the biggest cost factors associated with restoration plantings – in terms of both initial setup and ongoing maintenance (Freudenberger *et al.* 2004). Additionally, plants often require care in the initial stages to prevent seedling mortality due to water shortages or damage by grazing or weather. For financial reasons, it is often most feasible to implement smaller rather than larger plantings, despite the majority of literature recommending larger plantings to maximise biodiversity benefits (Lindenmayer *et al.* 2018b).

Importance of ensuring success

It is vital to assess the effectiveness of restoration plantings as a conservation strategy in fragmented agricultural landscapes. This is due to both the costs involved, and the reliance on restoration plantings for wildlife conservation in the absence of other landscape-scale

conservation strategies. There is little point spending vast sums of money creating habitat that does not function to conserve woodland avifauna, or at worst, exacerbates local population declines (e.g. by creating ecological traps; see Battin 2004). Furthermore, conservation funding is tight, with environmental budgets consistently reduced by governments at the local, state, and federal levels. It is crucial to assess whether the implementation of restoration plantings is achieving fundamental conservation goals – to restore functional woodland habitat and support woodland fauna.

Focus and limitations of current research

Multiple studies have investigated how woodland birds respond to restoration plantings. These are reviewed and summarised in Chapter 1. To date, an overwhelming majority of the research into restoration plantings as habitat for birds has focused on pattern data, using point counts and species diversity indices to make inferences about habitat quality. However, this approach relies on a number of assumptions. The presence of a species at a site does not necessarily mean that the site can support a breeding population of that species, or that it contains a resident population. Traditional bird survey data may therefore mask fundamental problems pertaining to the suitability and quality of habitat (Schlaepfer *et al.* 2002; Battin 2004). Restored habitat patches should support resident populations of bird species, which remain over multiple years, persist over the course of the breeding season, and are able to successfully reproduce (Barrett *et al.* 2008; Selwood *et al.* 2009). The persistence mechanisms of woodland birds are largely undocumented (Sunnucks 2011), and very little research has explored whether birds can actually survive and persist in revegetated areas. The suitability of restoration plantings as long-term habitat for woodland birds therefore remains in question.

The research described in this thesis is innovative in its exploration of this significant knowledge gap, and builds on long-term research by Lindenmayer *et al.* to further investigate the role of restoration plantings in supporting viable populations of woodland birds. Previous studies have compared the breeding success of birds in fragmented vs. intact habitats (e.g. Vander Haegen *et al.* 2002; Luck 2003), and Selwood *et al.* (2009) examined bird breeding activity in restoration plantings of varying age. Barrett *et al.* (2008) reported on a banding study monitoring the persistence of woodland birds in new restoration plantings, using the presence of a brood patch as an indicator of breeding activity. However, to my knowledge, there have been no published studies with the primary aim of quantifying the breeding success and persistence of birds in restoration plantings. This thesis is novel in attempting to address questions that go beyond the scope of conventional point count bird surveys, using mechanistic indicators to explore in detail whether restoration plantings can indeed support bird populations in the long term.

The knowledge gained from in-depth studies such as the one reported here is crucial to be able to effectively evaluate management outcomes as well as inform future restoration efforts (Block *et al.* 2001; Bennett and Watson 2011). The danger in failing to identify the mechanisms of species persistence in restored landscapes can be twofold: (a) mistakenly choosing the wrong types of restoration to establish due to misleading information on restoration effectiveness, and (b) wasteful investment in managing the wrong locations when resources for biodiversity conservation are already limited (A. Tulloch pers. comm. 2015). At worst, thinking a restoration action is improving population processes such as breeding success when it is actually only providing marginal habitat that birds occupy on a temporary basis, could lead to local declines and even extinctions of species that fail to reproduce in

revegetated patches (Kokko and Sutherland 2001; Battin 2004). These issues have so far not been adequately addressed in the research on restoration.

Aim and scope

Overarching research question

The broad aim of the research presented in this thesis was to assess whether restoration plantings are able to support resident, breeding populations of woodland birds. This aim is placed in the context of a landscape that has been heavily modified, with little of the original woodland habitat remaining. A push to restore habitat in temperate woodland regions has led to the establishment of restoration plantings of varying size and shape. Woodland birds are known to occupy restoration plantings, and some species may even preferentially occupy plantings over woodland remnants (Lindenmayer *et al.* 2010b). However, little is known as to whether restoration plantings can support successful breeding by woodland birds, or whether birds are resident in these patches over time.

Approach

To determine whether restoration plantings provide suitable habitat for supporting resident woodland bird populations in the long-term, this study examines how breeding activity and nesting success in plantings compare to that in remnant woodland patches, and explores the effects of different planting characteristics on these variables. The study also examines the territory sizes of individual birds to assess resource availability within plantings – larger territory sizes are typically associated with poorer quality habitat, as birds must forage over a greater area to obtain sufficient food resources (Zanette *et al.* 2000). Finally, the study assesses the site fidelity of individual birds over multiple years – another useful indicator of

habitat quality. Examining bird populations in detail also provides an indication of annual turnover due to emigration and/or mortality.

Breeding activity

I first examined whether woodland birds were displaying evidence of breeding in restoration plantings and remnant woodland patches in farming landscapes. I undertook breeding activity surveys, employing a scoring system to rank different breeding behaviours (or other indicators of breeding such as the presence of old nests) according to how strongly they indicate breeding success (Mac Nally 2007). Although this index is only a proxy for breeding success, it enables a rapid assessment of whether birds are breeding in a particular site. It also facilitates the collection of data on species of conservation concern – these are rare in the landscape, and it can be extremely challenging to find and monitor sufficient nests to draw statistically robust conclusions. Scoring breeding activity is a useful first step in going beyond pattern data to a more detailed, population-oriented approach to measuring the success of restoration plantings as a conservation strategy.

Nesting success

To quantify breeding success, I monitored individual bird nests, tracking their survival over time and recording their fate (i.e. succeed or fail) and daily survival rate (DSR). Over two study seasons, I monitored a total of 222 nests constructed by 24 bird species. These included several species of conservation concern, such as the hooded robin, diamond firetail, and yellow-rumped thornbill. I collected sufficient data on two species – superb fairywren and willie wagtail – to analyse these species separately. I assessed habitat features around nest sites, taking detailed measurements of the nest site (e.g. height above ground, distance to patch edge, substrate, concealment), as well as the nest site surrounds (e.g. shrub cover,

ground layer composition, and floristic diversity). I used generalised linear mixed regression modelling to examine whether temporal variables, patch-level variables, nest-site variables, or microhabitat variables were better predictors of nest success for woodland bird species. Due to inherent differences in success rates for different nest types, I analysed dome-nesters and cup-nesters separately.

For this study, I elected not to conduct an artificial nest experiment to examine predation rates or to further investigate key predators in the study area. This decision was based on frequent discussions in the literature suggesting that artificial nest studies are limited in their reflection of natural nest predation events and the rates at which these occur (Haskell 1995; Wilson *et al.* 1998; Weidinger 2001; Zanette 2002; Berry and Lill 2003; Burke *et al.* 2004; Thompson and Burhans 2004; Haff and Magrath 2011; Fulton 2018). Of particular concern were findings documenting an inability of artificial nests to accurately estimate levels of predation by snakes (Davison and Bollinger 2000; Thompson and Burhans 2004), since snakes were thought to be key nest-predators in the South-west Slopes bioregion (D. Lindenmayer, pers. comm. 2015). In an extensive review of artificial nest experiments, Major and Kendal (1996) recommended that "…*conclusions about nesting ecology derived from the use of artificial nest experiments should be treated as preliminary*", yet researchers in the field have since continued to rely heavily on this experimental approach. In this study, I instead monitored nest predation using fixed wildlife cameras situated near real nests. Predation rates on natural nests should not be inadvertently increased by the presence of a researcher or wildlife camera (Stake and Cimprich 2003; Ibáñez-Álamo *et al.* 2012).

Site fidelity and home ranges

At each study site, I trapped woodland birds and fitted particular species with coloured leg bands. These colour-bands enabled individual identification of birds. When selecting species to colour-band, I chose species of conservation concern that were present in the greatest numbers in my selected study sites, along with more common species that were members of the insectivorous ground-foraging guild (Table 1). This decision was based on evidence that insectivorous, ground-foraging birds are the most susceptible to population decline, and members of this guild have shown the strongest trends of decline (Antos and Bennett 2006; Barrett *et al.* 2007; Razeng and Watson 2012).

Table 1 Woodland bird species tagged with individually distinguishable combinations of coloured leg-bands. Conservation status abbreviations: LC = Least Concern. CC = of Conservation Concern. V = listed as Vulnerable in NSW.

Common name	Species	Cons. status	Diet	Foraging strata	No. colour-banded
superb fairywren	*Malurus cyaneus*	LC	Insectivorous	Ground	344
yellow-rumped thornbill	*Acanthiza chrysorrhoa*	CC	Insectivorous	Ground	145
buff-rumped thornbill	*Acanthiza reguloides*	CC	Insectivorous	Mixed	32
grey shrikethrush	*Colluricincla harmonica*	LC	Insectivorous	Mixed	30
rufous whistler	*Pachycephala rufiventris*	CC	Insectivorous	Arboreal	16
white-browed babbler	*Pomatostomus superciliosus*	LC	Insectivorous	Ground	15
willie wagtail	*Rhipidura leucophrys*	LC	Insectivorous	Ground	12
red-capped robin	*Petroica goodenovii*	CC	Insectivorous	Ground	10
diamond firetail	*Stagonopleura guttata*	V	Granivorous	Ground	8
speckled warbler	*Pyrrholaemus sagittatus*	V	Insectivorous	Ground	6

With the assistance of volunteers, I mapped territories (or home ranges) of colour-banded birds during the breeding season. Home range and territory size (in the absence of density dependence; Flockhart *et al.* 2016) are important indicators of resource availability and distribution within a habitat patch (Zanette *et al.* 2000). Home range size in a patchy environment is known to be inversely related to maximum resource density and resource renewal rate, and the shape of the home range is also indicative of resource distribution within

the landscape (Ford 1983). I conducted a pilot study for home range size in 2015, using the superb fairywren and willie wagtail (Appendix E).

In my study, I elected not to band honeyeaters. I made this decision based on the high numbers of honeyeaters caught in the first two sites in which I banded – I was concerned that I would not have sufficient time or bands available to process all other species of interest. Further, based on the knowledge that many honeyeaters are seasonally migratory or partially nomadic (Keast 1968), I considered it unlikely that recapture rates for honeyeaters would be high enough to warrant the additional banding effort.

Study area

South-west Slopes bioregion

The South-west Slopes bioregion of NSW comprises 8,070,608 ha of intensively cleared box-gum grassy woodland in the Murrumbidgee Catchment area (NSW Office of Environment and Heritage 2016). It is an area of undulating ranges on the western slope of the Great Dividing Range. The land on which this study took place was traditionally owned and managed by the Wiradjuri people. It is now part of Australia's sheep-wheat belt, with the majority of arable land dedicated to the production of sheep, wheat, and beef cattle. Remnant vegetation consists of patches of box-gum grassy woodland and derived native grassland, and is predominantly Blakely's red gum *Eucalyptus blakelyi* / white box *E. albens* / yellow box *E. melliodora* grassy woodland, a critically-endangered ecological community. Patches of box-ironbark woodland (mugga ironbark *E. sideroxylon*) and red stringybark woodland (red stringybark *E. macrorhyncha*) exist in some areas.

Climate

The climate is sub-humid, with no dry season; rainfall events occur at any time of year (NSW Office of Environment and Heritage 2016). Summers are hot, and winters are cool to cold with regular frosts. Annual rainfall is 360-1266 mm (NSW Office of Environment and Heritage 2016). My two major field seasons were conducted in years of above average rainfall (Bureau of Meteorology 2019).

Avifauna

There are 265 native and 12 introduced bird species that occur within the South-west Slopes bioregion. These include a diversity of honeyeaters (Meliphagidae), thornbills and allies (Acanthizidae), whistlers (Pachycephalidae), butcherbirds and woodswallows (Artamidae), parrots (Psittaculidae), raptors (Accipitridae and Falconidae), and fairywrens (Maluridae). Commonly encountered species include the superb fairywren, willie wagtail, grey fantail (*Rhipidura albiscapa*), yellow-rumped thornbill, yellow thornbill (*Acanthiza nana*), striated pardalote (*Pardalotus striatus*), red-capped robin, grey shrikethrush (*Colluricincla harmonica*), and weebill.

Seasonal migrants that arrive from northern and inland Australia to breed in spring/summer include the rufous songlark (*Cincloramphus mathewsi*), rufous whistler, sacred kingfisher (*Todiramphus sanctus*), and white-winged triller (*Lalage tricolor*). Altitudinal migrants such as the scarlet robin, flame robin and golden whistler (*Pachycephala pectoralis*) are present in the South-west Slopes during the winter months, but retreat to higher elevations in spring/summer for the breeding season.

The bird assemblage of the South-west Slopes bioregion is undergoing changes that reflect the ongoing loss and degradation of woodland habitat. Woodland-dependent species are experiencing population declines, while bird species that prefer or tolerate open, fragmented habitat have become more dominant (Barrett *et al.* 2007; Ford 2011). Degradation and fragmentation of woodlands also favour the noisy miner, a hyperaggressive honeyeater that competes with and excludes other small woodland birds (Maron 2007; Bennett *et al.* 2015b; Beggs *et al.* 2019). The South-west Slopes is home to at least 15 woodland bird species listed as either Vulnerable or Endangered under the NSW Threatened Species Conservation Act 1995, including the crested shriketit, grey-crowned babbler, speckled warbler, hooded robin, scarlet robin, flame robin, varied sittella, brown treecreeper, diamond firetail, black-chinned honeyeater, dusky woodswallow (*Artamus cyanopterus*), little lorikeet (*Glossopsitta pusilla*), swift parrot (*Lathamus discolor*), superb parrot (*Polytelis swainsonii*), and turquoise parrot (*Neophema pulchella*) (NSW OEH 2019).

Scope of the research

"Woodland birds"

Woodland birds have suffered significant declines since European settlement, but what do we actually mean when we refer to "woodland birds"? For the purposes of this thesis, a "woodland bird" was defined as a small to medium-sized terrestrial bird species that is typically found in a box-gum grassy woodland community in south-eastern Australia. In Chapter 2 (breeding activity), I refer to the collective suite of species as the "woodland assemblage". For the remainder of papers in this study, I excluded larger predatory species such as magpies and ravens, and focused on small to medium-sized woodland bird species. Species of conservation concern are defined as species that appear on threatened species lists at a federal or state level, and species that declined by >20% in the South-west Slopes

bioregion between the first and second *Atlas of Australian Birds* (Barrett *et al.* 2003; Barrett *et al.* 2007).

Species of conservation concern accounted for a relatively small proportion of the data collected in this study. This is consistent with the theory of beta diversity in ecological communities – the majority of species in any given assemblage are rare, and assemblages tend to be dominated by a small number of common species (Wilson and Shmida 1984; Koleff *et al.* 2003). Ignoring dominant species in an attempt to focus on species of conservation concern not only limits the scope of data that can be collected, but also risks overlooking community dynamics that are influenced by common or abundant species. For this reason, I decided to select several "common" species for my study on site fidelity and home ranges.

"Reference sites"

A key element of my study was the comparison of restoration plantings and similarly sized remnant patches (1.3–7.8 ha) with larger "reference sites". These reference sites were much larger patches of remnant box-gum grassy woodland, ranging in size from around 47 ha to approximately 110 ha, and were used to represent "intact" woodland in the study region. Other studies in temperate woodlands have used different criteria for large and/or intact remnant patches. For example, Zanette *et al.* (2000) defined small patches as ~55 ha, and large patches as >400 ha. Similarly, small patches defined by Major and Gowing (2002) were 40–105 ha, and large patches were >500 ha. Contiguous forest areas used as reference sites by Burke and Nol (2000) were >10,000 ha. Site selection for my study was constrained by the lack of large remnant patches in the South-west Slopes bioregion to serve as reference sites. It is important to recognise that the largest remnant patches in this landscape may not accurately reflect ecological processes in truly intact woodland.

Problem statement/research gap

Ecological traps

If birds are occupying restoration plantings but failing to breed successfully, then it is possible that plantings are acting as ecological traps in fragmented agricultural landscapes. Ecological traps occur when birds use misleading habitat cues to disproportionately colonise sites that are unsuitable for breeding or long-term survival (Battin 2004). For a site to support breeding and long-term survival, it must provide the resources that a bird requires to sustain its daily activities and to reproduce. These resources include food, nesting sites, shelter from predators and other threats, and social and/or mating opportunities.

If restoration plantings are failing to support breeding birds (or even resident birds), and are either functioning as temporary refugia while birds are moving around the landscape in search of something more suitable, or acting as ecological traps for birds that attempt to breed in them, then restoration plantings could at worst exacerbate rather than address population declines.

Significance/contribution

This study is novel in its use of new metrics by which to assess the effectiveness of restoration plantings and other conservation strategies for wildlife. I argue that it is insufficient to assess the success of a newly created patch of habitat simply by surveying for presence/abundance of species at the site. This information can provide valuable insights and is certainly a good starting point, but only provides a limited picture of how fauna are using the site. “Success” of wildlife conservation strategies should be contingent on the presence of resident, self-sustaining populations of key fauna species.

The contributions of this thesis are largely in applied conservation. Through my work, I have addressed key questions pertaining to the ways in which woodland birds respond to restoration plantings. My recommendations can be used to improve the quality of restoration plantings, design efficient restoration programs, and ensure that plantings are cost-effective as a conservation strategy. Ultimately, my research provides valuable insights into how to better support declining and threatened woodland bird species.

Chapter outlines

Chapter 1: Literature review

The first chapter of this thesis is a literature review that synthesises current knowledge of birds in restoration plantings and fragmented agricultural landscapes. I identify key knowledge gaps and propose future directions for research pertaining to woodland bird conservation in agricultural landscapes. I compile a set of specific research questions that should be addressed as priorities. Several of these questions form the basis of this thesis. This chapter has been published as a review article in *Wildlife Research*.

Chapter 2: Breeding activity

My second chapter assesses evidence of breeding activity by birds in restoration plantings and similarly sized woodland remnants. I undertook breeding activity surveys in the peak breeding season for birds in the South-west Slopes bioregion (September to November). I employed a scoring system developed by Mac Nally (2007) that ranks indicators of breeding behaviour according to how strongly they indicate breeding success. For example, observing a bird carrying nesting material scores a 3.0, while observing a nest with nestlings scores a 6.0. Scores are then tallied for a score per site or per survey and modelled against explanatory variables. This chapter focused on site variables, including patch type, size, and shape. I also

included patch age and presence of fences as two additional explanatory variables. I used linear mixed effects regression modelling to examine the effects of my variables of interest on the response variable. In this chapter, I also compare the relative abundance of birds in my study sites with their breeding activity scores. This chapter has been published as a research article in *Biological Conservation*.

Chapter 3: Nesting success

In my third chapter, I take a more detailed approach to measuring breeding success. During the breeding activity surveys described in Chapter 2, I marked the locations of all nests observed during the surveys. I then tracked the fate of nests using a combination of in-person visits and remote wildlife cameras. I used two different indicators of nest success: nest fate (succeed or fail), and daily nest survival (DSR). DSR represents the daily probability of a nest surviving over the course of the nesting period, and enables comparison between nests encountered at different "ages". I modelled nest success and DSR against patch attributes (site type, size, shape), nest-site variables (concealment, height, distance to patch edge), temporal variables (year, date of discovery), and microhabitat variables around the nest site (ground layer composition and shrub cover). Additionally, I used motion-sensing wildlife cameras to identify the key nest-predators in my study sites. This chapter has been published as a Highlighted Student Research article in *Oecologia*.

Chapter 4: Survival, site fidelity and home ranges

My final chapter examines in detail the responses of individual birds to habitat patches (restoration plantings and woodland remnants) in fragmented agricultural landscapes. I banded over 1000 woodland birds, and conducted recapture efforts in subsequent years to estimate annual survival and site fidelity. I also colour-banded several species to facilitate

resighting of individuals without the need to physically recapture them. These included species of conservation concern such as the diamond firetail and yellow-rumped thornbill. Very few Australian studies have attempted to monitor survival or site fidelity of woodland birds. To my knowledge, this is only the second study to undertake bird banding in restoration plantings (see Barrett *et al.* 2008), and the first study to undertake banding in established plantings of more than ten years of age.

I also tracked the home ranges of individual woodland birds to document landscape-scale habitat-use in restoration plantings and woodland remnants. I selected the superb fairywren and willie wagtail as the two target species for this study, as they were relatively easy to locate and track, and found in the majority of my study sites. With the assistance of volunteers, I collected home range data by tracking birds on foot with a handheld GPS. I recorded their locations, the habitat features they used, and the activities they performed. Although this method is labour-intensive compared to GPS tracking, it is impossible to fit GPS trackers on many small bird species without exceeding the ethical weight limit of 5% of their bodyweight (Caccamise and Hedin 1985). Radio tracking was also unsuitable for this study, as the level of precision obtained via radio tracking is insufficient to accurately capture the movements of woodland birds over short distances. Tracking birds on foot facilitates the collection of detailed data on habitat use and behaviour.

To my knowledge, no prior studies have attempted to assess the habitat-use of birds in restoration plantings, and very few studies of this nature have been undertaken on woodland birds in general. My study provides valuable insights into the habitat-use of woodland birds in restoration plantings and woodland remnants, details interesting aspects of home range size and shape, and documents site fidelity of several species. Additionally, I provide the first

evidence of site fidelity by a seasonally migratory bird species, the rufous whistler, in restoration plantings. This chapter was submitted to *Ecological Applications* in November 2019, and at the time of thesis submission, had been resubmitted to the same journal following revisions that were recommended by anonymous reviewers.

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CHAPTER 1

Beyond pattern to process: current themes and future directions for the conservation of woodland birds through restoration plantings

Weebill (*Smicrornis brevirostris*) captured in a restoration planting in the South-west Slopes bioregion, NSW. Photo: Donna Belder.

Belder D. J., Pierson J. C., Ikin K., and Lindenmayer D. B. (2018). Beyond pattern to process: Current themes and future directions for the conservation of woodland birds through restoration plantings. *Wildlife Research* **45**, 473–489.

Abstract

Habitat loss as a result of land conversion for agriculture is a leading cause of global biodiversity loss and altered ecosystem processes. Restoration plantings are an increasingly common strategy to address habitat loss in fragmented agricultural landscapes. However, the capacity of restoration plantings to support reproducing populations of native plants and animals is rarely measured or monitored. This review focuses on avifaunal response to revegetation in Australian temperate woodlands, one of the world's most heavily altered biomes. Woodland birds are a species assemblage of conservation concern, but only limited research to date has gone beyond pattern data and occupancy trends to examine whether they persist and breed in restoration plantings. Moreover, habitat quality and resource availability, including food, nesting sites and adequate protection from predation, remain largely unquantified. Several studies have found that some bird species, including species of conservation concern, will preferentially occupy restoration plantings relative to remnant woodland patches. However, detailed empirical research to verify long-term population growth, colonisation and extinction dynamics is lacking. If restoration plantings are preferentially occupied but fail to provide sufficient quality habitat for woodland birds to form breeding populations, they may act as ecological traps, exacerbating population declines. Monitoring breeding success and site fidelity are under-utilised pathways to understanding which, if any, bird species are being supported by restoration plantings in the long term. There has been limited research on these topics internationally, and almost none in Australian temperate woodland systems. Key knowledge gaps centre on provision of food resources, formation of optimal foraging patterns, nest-predation levels and the prevalence of primary predators, the role of brood parasitism, and the effects of patch size and isolation on resource availability and population dynamics in a restoration context. To ensure that restoration plantings benefit woodland birds and are cost-effective as conservation strategies, the

knowledge gaps identified by this review should be investigated as priorities in future research.

Additional keywords: breeding success, population dynamics, revegetation.

Introduction

A large fraction of the world's woodland and forest avifauna is declining (IUCN 2016; Waldron *et al*. 2017), reflecting the well-documented global trend of biodiversity loss associated with intensifying anthropogenic activities (Butchart *et al*. 2010). An increasingly common strategy to address habitat loss in fragmented agricultural landscapes is the creation of habitat through revegetation, often referred to as 'restoration plantings' (Pastorok *et al*. 1997; Cairns 2000; Rey Benayas *et al*. 2009; Barral *et al*. 2015). These are typically small patches of planted native vegetation, and are often intended to facilitate landscape connectivity and conservation of fauna such as birds (Block *et al*. 2001; Freudenberger 2001).

Patterns of bird species occupancy and abundance in restoration plantings are commonly used to infer habitat quality (Cunningham *et al*. 2008; Munro *et al*. 2011; Lindenmayer *et al*. 2012). However, there has been limited research on the population responses of birds to restoration plantings or other forms of habitat restoration, such as remediation (Larison *et al*. 2001; Germaine and Germaine 2002). It is crucial to understand the population dynamics of birds in revegetated landscapes to establish whether restoration plantings provide quality habitat in which birds can survive and reproduce. This is particularly relevant for threatened and declining bird assemblages that may come to rely on restoration plantings for long-term population stability.

The ecological value of temperate woodland restoration plantings for woodland birds in Australia has traditionally been assessed using pattern data, primarily presence and abundance of bird species in study sites. This pattern-based research (e.g. Table 1.1) provides a critical basis for understanding the potential value of restoration plantings for woodland birds in fragmented environments. However, to supplement the existing body of knowledge, a much

deeper understanding is needed of the demographic and behavioural responses (e.g. survival, site fidelity, breeding success, dispersal) of woodland bird populations to habitat restoration. This is fundamental to determine the conservation and management value of restoration plantings, including their potential contribution to reversing species declines (Bennett and Watson 2011). For example, species that have been classified as 'planting specialists' (Table 1.1) may be expected to successfully breed in restoration plantings, but this has not been adequately tested. It is, therefore, essential to begin to explore these processes in a restoration context, asking the following question: 'Do restoration plantings facilitate the long-term persistence of birds in fragmented landscapes?'.

Previous research on bird community population dynamics, such as breeding success, has mostly dealt with birds in remnant habitat (e.g. Hoover *et al*. 1995; Zanette and Jenkins 2000; Berry 2001; Zanette 2001; Herkert *et al*. 2003; Debus 2006*a*, 2006*b*; Holoubek and Jensen 2016), with a subset of comparative studies in fragmented and intact landscapes (e.g. Burke and Nol 2000; Cooper *et al*. 2002; Luck 2003). The majority of earlier work in revegetated landscapes has focused on species richness and abundance, with an emphasis on monitoring for occupancy by birds through time after establishment of restoration plantings (e.g. Taws 2002; Twedt *et al*. 2002; Martin *et al*. 2004; Barrett *et al*. 2008; Saunders and Nicholls 2008; Freeman *et al*. 2009; Gould 2011; Munro *et al*. 2011; Becker *et al*. 2013; Lindenmayer *et al*. 2016).

This earlier research has collectively established that some woodland bird species are able to colonise and occupy restoration plantings. The pressure of potential extinction debts for woodland birds (Ford *et al*. 2009), that is, continued declines even after habitat loss and degradation (or other challenges) are eliminated or reversed (Kuussaari *et al*. 2009), adds

impetus to the need for replacing lost woodland habitat. However, it is imperative the effects of revegetation on avifauna are more comprehensively understood, lest they fail to address (or at worst, exacerbate) population declines.

Approach

In the present paper, we review the current knowledge on avifaunal response to revegetation and habitat restoration, and provide a general overview and synthesis of existing and future research directions on the topic of woodland birds in restoration plantings. We focus largely on Australian temperate woodlands, the cover of which has been reduced by up to 90% over the past 150 years as a result of land clearing for agriculture (Paton and O'Connor 2010). We build on the preliminary overview by Munro *et al*. (2007), consolidating the most recent research on the relationship between birds and restoration plantings and examining the available information that underpins practical restoration of woodland habitat. We move beyond the scope of previous reviews by exploring how the implementation of restoration plantings might influence the long-term survival and persistence of woodland bird communities in fragmented agricultural landscapes. Finally, we identify gaps in the current knowledge and propose further research that would enhance understanding of the population dynamics of woodland birds in restoration plantings and revegetated landscapes.

We identified relevant literature for the present paper by searching publication databases and citation lists, including ScienceDirect, Scopus and Google Scholar. We took a non-systematic approach and used a broad range and combination of search terms, including 'woodland birds', 'breeding success', 'population dynamics', 'occupancy', 'distribution', 'revegetation' and 'restoration'. We searched the internet and an institutional library catalogue for non-peer-reviewed work, including books, theses and reports.

Background

Habitat degradation and restoration

Temperate woodlands once covered an extensive area of southern Australia, however, most have been cleared for agriculture since European settlement (Saunders and Curry 1990; Lindenmayer *et al*. 2010*a*; Bradshaw 2012). Estimates vary, but ~32million hectares, or up to 90%, of native temperate woodland vegetation cover has been cleared (Vesk and Mac Nally 2006; Paton and O'Connor 2010). Scattered remnants persist, but because of their isolation and degradation history, they are vulnerable to threatening processes such as agricultural intensification, grazing, nutrient enrichment, weed invasion and climate change (Eldridge 2003; Maron and Fitzsimons 2007; Duncan and Dorrough 2009; Mac Nally *et al*. 2009; Prober *et al*. 2012, 2014).

The negative effects of broad-scale habitat clearance on the Australian environment began to be widely recognised in the 1980s (Saunders *et al*. 1991; Hobbs and Saunders 2012; Lindenmayer *et al*. 2013; Campbell *et al*. 2017). Changes in attitude towards land management throughout the 1980s and 1990s led to small-scale revegetation programs that were initially instigated by the farming and environmental sectors to address issues such as salinity and erosion (Stirzaker *et al*. 2002; Campbell *et al*. 2017), with larger-scale government- initiated revegetation programs such as the National Tree Program and the One Billion Trees Program applied within the next two decades (Hajkowicz 2009; Lindenmayer *et al*. 2013). Many early plantings were implemented without a well-defined wildlife conservation plan, but have, nonetheless, in some cases been occupied by woodland birds and other fauna (Munro *et al*. 2007; Lindenmayer *et al*. 2016).

In more recent years, some restoration plantings have been implemented with clear plans and goals relating to ecological factors, such as the habitat requirements of focal species (Freudenberger 2001; Lindenmayer *et al*. 2013). Knowledge of effective revegetation techniques has also been used to begin construction of large-scale habitat-linkage corridors (e.g. Gondwana Link) through the acquisition and revegetation of farming properties (Paton and O'Connor 2010). An ongoing (up to 2020), large-scale government initiative is the 20 Million Trees Program (Department of the Environment and Energy 2017), which aims to 'improve the extent, connectivity and condition of native vegetation', with explicit reference to threatened species such as the southern emu-wren (*Stipiturus malachurus*) and regent parrot (*Polytelis anthopeplus*) (Department of the Environment and Energy 2017; Landcare Australia 2017). Vegetation is also increasingly being planted for carbon sequestration, and such plantings have the potential to enhance the conservation of biodiversity (Bradshaw *et al*. 2013; Collard *et al*. 2013).

With ongoing large-scale revegetation programs such as the 20 Million Trees Program underway in Australia, extensive areas of temperate woodland restoration plantings are being added to the landscape every year (Atyeo and Thackway 2009; Campbell *et al*. 2017). However, it is important to note that Australia's rate of land clearing remains among the highest in the world (Bradshaw 2012; Evans 2016). With an ongoing net loss of habitat, restoration plantings are a critical conservation strategy for woodland birds and other fauna. Many restoration projects claim to focus on creating habitat for threatened or declining wildlife (e.g. Landcare Australia 2017). There is evidence that a focal-species approach can be used to develop guidelines for revegetation programs (Freudenberger 2001; Freudenberger and Brooker 2004; Wood *et al*. 2004). However, its usefulness as a conservation tool is debated (Lambeck 2002; Lindenmayer *et al*. 2002). Recent research suggests that although

the focal-species approach has some merit, it is also necessary to ensure the flexibility of management actions such that all species are accounted for in conservation; focusing on one species may not benefit others of conservation concern, especially those that might not occur in species-rich assemblages (Lindenmayer *et al*. 2014). Furthermore, a generalised lack of information on the habitat requirements and population processes of many threatened and declining woodland bird species (Rayner *et al*. 2014) means that many revegetation programs are being implemented without sufficient knowledge as to the habitat requirements of the species they should be supporting (Block *et al*. 2001; Montague-Drake *et al*. 2009; Polyakov *et al*. 2015).

Reviews of restoration practice as early as the 1990s have outlined steps that should be taken to ensure the successful restoration of fragmented and degraded ecosystems, as well as challenges posed by large-scale revegetation (Pastorok *et al*. 1997; Block *et al*. 2001; Hobbs 2003; Lindenmayer *et al*. 2008; Duncan and Dorrough 2009; Prober and Smith 2009; Campbell *et al*. 2017); also see the National Standards for the Practice of Ecological Restoration in Australia (McDonald *et al*. 2016). The importance of setting measurable goals for restoration is crucial and underpins how we define long-term success in a restoration context (Cairns 2000; Block *et al*. 2001; Ruiz-Jaen and Aide 2005; Herrick *et al*. 2006; Hobbs 2017). This should include assessing the capacity of restoration plantings to support reproducing populations, an attribute that is rarely measured in restoration monitoring projects (Ruiz-Jaen and Aide 2005; Vesk and Mac Nally 2006).

Patterns: bird responses to revegetation in Australian temperate woodlands

Many pattern-based studies have investigated the effects of habitat loss, fragmentation and degradation on declining woodland bird species in Australia (reviewed by Ford *et al*. 2001; Ford 2011); fewer have examined how these species respond to restoration plantings (Nichols and Watkins 1984; Heath 2003; Robinson 2006; Lindenmayer *et al*. 2007, 2010*b*, 2012; Barrett *et al*. 2008; Cunningham *et al*. 2008; Saunders and Nicholls 2008; Loyn *et al*. 2009; Selwood *et al*. 2009; Munro *et al*. 2011; Shanahan *et al*. 2011; Bennett *et al*. 2013; Vesk *et al*. 2015). Much of the research on birds in revegetated landscapes has focused on answering the question 'Do birds use restoration plantings?', and, concurrently, 'Which plantings are preferentially selected?'.

Previous research has discovered that some woodland bird species, including species of conservation concern, will readily occupy restoration plantings, and may even preferentially select plantings over remnant woodland (Nichols and Watkins 1984; Heath 2003; Kinross 2004; Martin *et al*. 2004; Kavanagh *et al*. 2007; Cunningham *et al*. 2008; Saunders and Nicholls 2008; Loyn *et al*. 2009; Lindenmayer *et al*. 2010*b*, 2012; Martin *et al*. 2011). These species have been termed 'planting specialists', that is, species that are more likely to be found in restoration plantings than in woodland remnants (Table 1.1). It should be noted that inferred habitat preferences for some species, such as the eastern yellow robin, scarlet robin, and southern whiteface (see Table 1.1 for scientific names), are not consistent among studies.

Table 1.1 Planting specialists
Woodland bird species identified as 'planting specialists' – bird species more likely to be found in plantings than in remnants or other sites – in Australian studies of bird occurrence, distribution and abundance in revegetated landscapes. Species are listed in taxonomic order (Christidis and Boles 2008).

Species		Studies	Study region(s)
superb fairy-wren	*Malurus cyaneus*	Barrett *et al.* 2008; Cunningham *et al.* 2008; Martin *et al.* 2011; Lindenmayer *et al.* 2012	South-west Slopes, NSW
white-browed scrubwren	*Sericornis frontalis*	Cunningham *et al.* 2008	South-west Slopes, NSW
speckled warbler[C]	*Chthonicola sagittata*	Kavanagh *et al.* 2007; Cunningham *et al.* 2008; Lindenmayer *et al.* 2012	South-west Slopes, NSW
weebill[C]	*Smicrornis brevirostris*	Kavanagh *et al.* 2007; Cunningham *et al.* 2008; Martin *et al.* 2011	South-west Slopes, NSW
western gerygone	*Gerygone fusca*	Cunningham *et al.* 2008; Lindenmayer *et al.* 2012	South-west Slopes, NSW
striated thornbill	*Acanthiza lineata*	Kavanagh *et al.* 2007	South-west Slopes, NSW
yellow thornbill	*Acanthiza nana*	Kavanagh *et al.* 2007; Cunningham *et al.* 2008; Martin *et al.* 2011; Lindenmayer *et al.* 2012	South-west Slopes, NSW
yellow-rumped thornbill[C]	*Acanthiza chrysorrhoa*	Cunningham *et al.* 2008; Martin *et al.* 2011; Lindenmayer *et al.* 2012	South-west Slopes, NSW
southern whiteface[C]	*Aphelocephala leucopsis*	Barrett *et al.* 2008;	South-west Slopes, NSW
white-plumed honeyeater	*Ptilotula penicillata*	Barrett *et al.* 2008; Martin *et al.* 2011; Lindenmayer *et al.* 2012	South-west Slopes, NSW
red wattlebird	*Anthochaera carunculata*	Cunningham *et al.* 2008; Lindenmayer *et al.* 2012	South-west Slopes, NSW
rufous whistler[C]	*Pachycephala rufiventris*	Kavanagh *et al.* 2007; Lindenmayer *et al.* 2012	South-west Slopes, NSW
grey shrike-thrush	*Colluricincla harmonica*	Martin *et al.* 2011; Lindenmayer *et al.* 2012	South-west Slopes, NSW
grey fantail	*Rhipidura albiscapa*	Cunningham *et al.* 2008; Lindenmayer *et al.* 2012	South-west Slopes, NSW
willie wagtail	*Rhipidura leucophrys*	Heath 2003; Martin *et al.* 2011; Lindenmayer *et al.* 2012	Goomalling Shire, WA; South-west Slopes, NSW
scarlet robin[CV]	*Petroica boodang*	Cunningham *et al.* 2008	South-west Slopes, NSW
red-capped robin[C]	*Petroica goodenovii*	Cunningham *et al.* 2008; Lindenmayer *et al.* 2012	South-west Slopes, NSW
flame robin[CV]	*Petroica phoenicea*	Lindenmayer *et al.* 2012	South-west Slopes, NSW
hooded robin[CV]	*Melanodryas cucullata*	Cunningham *et al.* 2008	South-west Slopes, NSW
eastern yellow robin	*Eopsaltria australis*	Cunningham *et al.* 2008	South-west Slopes, NSW
red-browed finch	*Neochmia temporalis*	Kavanagh *et al.* 2007; Barrett *et al.* 2008; Cunningham *et al.* 2008; Lindenmayer *et al.* 2012	South-west Slopes, NSW
diamond firetail[CV]	*Stagonopleura guttata*	Cunningham *et al.* 2008	South-west Slopes, NSW

[C] Of conservation concern
[V] Classified as Vulnerable in NSW

Bird species occupancy and abundance in restoration plantings appear to be influenced by a complex relationship between context (location within the landscape, e.g. proximity to other areas of native vegetation), configuration (e.g. shape, area) and content (structural and floristic variables) (Nichols and Watkins 1984; Kavanagh *et al*. 2007; Cunningham *et al*. 2008; Kinross and Nicol 2008; Lindenmayer *et al*. 2010*b*, 2016; Munro *et al*. 2011; Table 1.2). Differences in bird community composition in restoration plantings and remnant woodland have been consistently reported in Australia (Arnold 2003; Loyn *et al*. 2007; Martin *et al*. 2011; Munro *et al*. 2011; Lindenmayer *et al*. 2012), as well as in similarly restored habitat patches in Brazil (Becker *et al*. 2013), China (Zhang *et al*. 2011), Mexico (MacGregor-Fors *et al*. 2010) and the United States (Brawn 2006; Ortega-Álvarez *et al*. 2013). Some studies have noted that the bird community continually changes following initial establishment as planted vegetation matures and becomes more similar to remnant habitat (Lindenmayer *et al*. 2016; Debus *et al*. 2017); generalists and species favoured by open habitats are more common in the early stages, whereas shrub-dwelling and canopy specialists colonise as the habitat structure develops over time (Twedt *et al*. 2002; Heath 2003; Jansen 2005; Freeman *et al*. 2009; Gould and Mackey 2015).

Habitat composition and structure strongly influence the composition and abundance of bird communities in restoration plantings (Arnold 2003; Barrett *et al*. 2008; Munro *et al*. 2011; Gould and Mackey 2015). In general, woodland bird abundance and diversity appear to increase with an increasing habitat complexity; the inclusion of a more diverse plant species assemblage, leaf litter, and an increase in canopy cover have all been positively associated with bird species richness and abundance (Barrett *et al*. 2008; Bonifacio *et al*. 2011; Munro *et al*. 2011; Gould and Mackey 2015). It is important to recognise the diverse ways in which different species or foraging guilds may respond to habitat features in restoration plantings.

For example, Comer and Wooller (2002) found that a 'clumped' spatial arrangement of shrubs in restoration plantings facilitated competitive exclusion of small honeyeaters by larger species, decreasing overall nectarivore diversity in the plantings. Barrett *et al*. (2008) found that ground-foraging insectivores were under-represented in restoration plantings, and postulated that lack of native forb diversity may have been a likely cause. According to Arnold (2003), the inclusion of canopy and perching sites within 1 m of the ground results in a greater abundance of insectivores in restoration plantings. Martin *et al*. (2004) found significantly lower abundances of species that primarily forage on bark in restoration plantings than in woodland remnants; this may be due, in part, to the fact that certain habitat features, such as decorticating bark and fallen timber, take decades or even centuries to develop in temperate woodland habitats (Cunningham *et al*. 2007; Mac Nally 2008; Vesk *et al*. 2008; Munro *et al*. 2009). This may also be why restoration plantings are not predicted to support certain woodland-dependent bird species until 40, 60, or 100 years after establishment (Thomson *et al*. 2009).

There is evidence that the amount and proximity of remnant or planted vegetation in the area surrounding a restoration planting may have as much, if not more, influence on bird assemblage than does the content of the planting itself (Kavanagh *et al*. 2007; Lindenmayer *et al*. 2007, 2010*b*). The rufous whistler (*Pachycephala rufiventris*) and grey fantail (*Rhipidura albiscapa*) are two species that exhibit a positive response to an increase in the amount of planted native vegetation surrounding a restoration planting (Lindenmayer *et al*. 2010*b*). A habitat patch that is close to other patches may provide better foraging opportunities for species with large home ranges, such as the rufous whistler. Well- connected restoration plantings may also be key to supporting species whose local persistence is limited by dispersal, such as the brown treecreeper (*Climacteris picumnus*).

Table 1.2 Restoration planting characteristics and woodland bird occupancy
Variables found to influence occupancy by bird species in restoration plantings in Australian studies of bird occurrence, distribution and abundance in revegetated landscapes. Adapted from Lindenmayer *et al.* (2010b).

Variable type	Variable	Studies	Study region(s)
Context	Landscape vegetation cover, distance to nearest other native vegetation	Heath 2003; Barrett *et al.* 2008; Selwood *et al.* 2009; Lindenmayer *et al.* 2010b; Munro *et al.* 2011	Goomalling Shire, WA; Box-ironbark region, VIC; South-west Slopes, NSW; West Gippsland, VIC
Configuration	Shape	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Area	Selwood *et al.* 2009; Lindenmayer *et al.* 2010b; Munro *et al.* 2011	Box-ironbark region, VIC; South-west Slopes, NSW; West Gippsland, VIC
	Topography	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
Content	No. plants	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	No. native plant species	Barrett *et al.* 2008; Munro *et al.* 2011	South-west Slopes, NSW; West Gippsland, VIC
	Canopy depth	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Canopy height	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Overstorey cover	Barrett *et al.* 2008; Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Midstorey cover	Barrett *et al.* 2008; Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Understorey/ground cover	Heath 2003; Arnold 2003; Barrett *et al.* 2008; Lindenmayer *et al.* 2010b	Goomalling Shire, WA; Wandoo woodland, WA; South-west Slopes, NSW
	Mistletoe	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Logs, fallen timber, leaf litter	Barrett *et al.* 2008; Selwood *et al.* 2009; Lindenmayer *et al.* 2010b; Munro *et al.* 2011	Box-ironbark region, VIC; South-west Slopes, NSW; West Gippsland, VIC
	Dead trees/shrubs	Lindenmayer *et al.* 2010b	South-west Slopes, NSW
	Remnant/paddock trees	Selwood *et al.* 2009; Lindenmayer *et al.* 2010b; Munro *et al.* 2011	Box-ironbark region, VIC; South-west Slopes, NSW; West Gippsland, VIC
	Grazing	Selwood *et al.* 2009; Lindenmayer *et al.* 2010b	Box-ironbark region, VIC; South-west Slopes, NSW
Other	Age	Selwood *et al.* 2009; Munro *et al.* 2011	Box-ironbark region, VIC; West Gippsland, VIC
	Vegetation condition	Munro *et al.* 2011	West Gippsland, VIC

Process: breeding and persistence in restoration plantings

Do restoration plantings actually provide suitable breeding habitat for woodland birds, and, if they do, are attempts at breeding by birds in these sites successful? To persist in the long term, birds must be able to gain required resources from the patch they select (or from adjacent areas). This includes resources such as food and nesting sites, but also habitat services such as adequate protection from predation and competition (Figure 1.1).

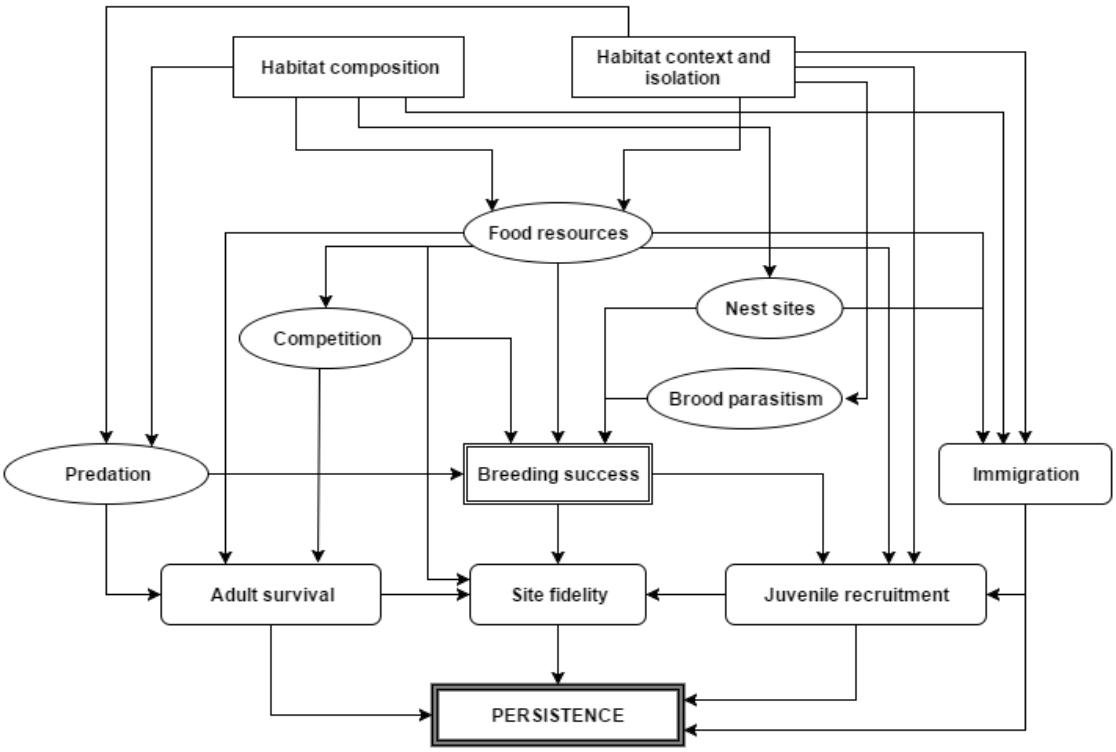

Figure 1.1 Conceptual diagram of interrelated factors that may influence the breeding success and persistence of woodland bird populations in restoration plantings. Bold/double rectangles = the processes we focus on in this review (breeding success and persistence). Rounded rectangles = population processes i.e. what the birds are doing. Rectangles = broad patch-level characteristics i.e. what type of habitat the birds are living in and where. Circles = fine-scale patch-level attributes i.e. what the birds experience in the habitat patch.

There is documented evidence of breeding activity and site fidelity in multiple woodland bird species colonising young restoration plantings (2–3 years old) (Barrett *et al*. 2008). Bird breeding activity also has been reported in more mature plantings (up to 26 years old for directly planted sites, and 111 years for restored woodland remnants) (Selwood *et al*. 2009;

Mac Nally *et al*. 2010; Bond 2011). However, species preference for, and occupancy of, a given habitat type is not necessarily correlated with long-term survival and persistence (Van Horne 1983; Battin 2004; Loyn *et al*. 2009). This is particularly relevant for declining species, which may occupy a site but display only limited evidence of successful breeding (Selwood *et al*. 2009; Mac Nally *et al*. 2010).

Restored habitats, including restoration plantings, have the potential to become ecological traps for bird populations. Ecological traps occur when individuals use habitat cues to preferentially colonise sites that are of inferior habitat quality or associated with lower breeding success than are other sites (Kokko and Sutherland 2001; Schlaepfer *et al*. 2002; Battin 2004; Robertson and Hutto 2006). This concept differs from an ecological 'sink', which is simply an area of poor-quality habitat that is not preferentially occupied, in which the population tends towards decline (Dias 1996). Individuals may also inadvertently avoid high-quality patches because of misleading habitat cues, which, likewise, creates an ecological trap mechanism at the landscape level (Gilroy and Sutherland 2007). If restoration plantings were to act as ecological traps, with remnant habitat patches as the population sources, metapopulation declines may be worsened rather than reversed by the extensive planting of native vegetation (Figure 1.2).

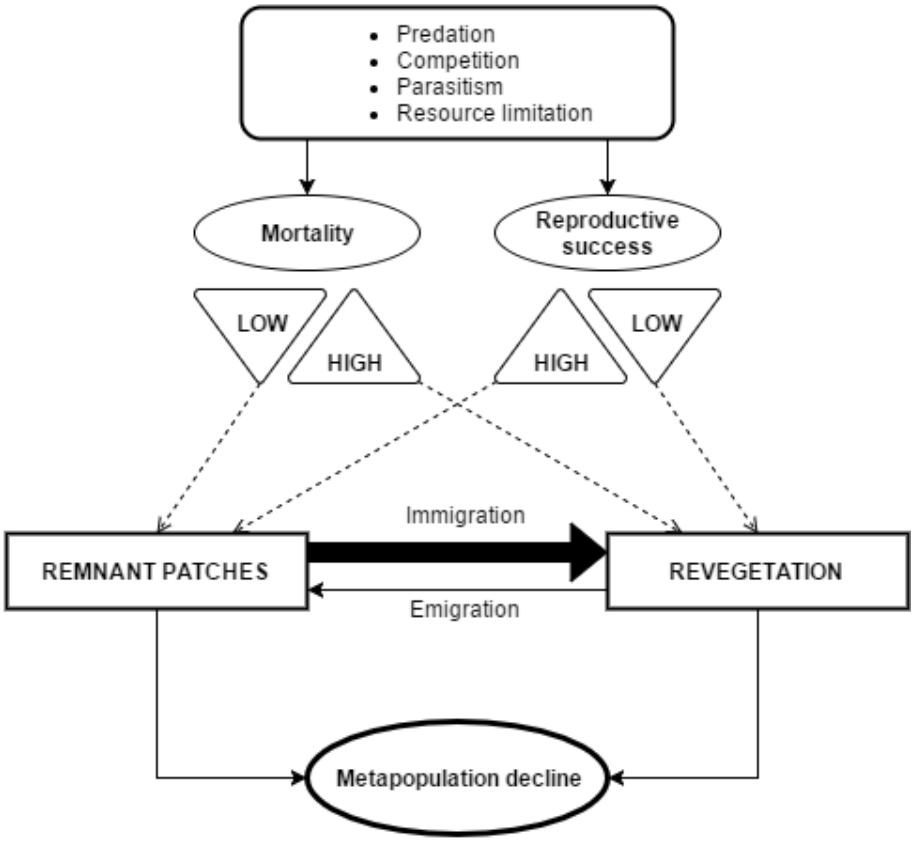

Figure 1.2 A conceptual model of an ecological trap mechanism operating in a fragmented landscape with restoration plantings and remnant patches. Restoration plantings have the potential to become ecological traps if they are preferentially occupied but lead to lower reproductive success and/or higher mortality than remnant patches. ○ = population process, △= trend in population process, □ = habitat type.

There are some instances in the global literature of restored habitats acting as ecological traps. For example, Larison *et al*. (2001) found that the song sparrow (*Melospiza melodia*) in California had lower reproductive success in restored riparian forest than it had in naturally regenerating or mature forest, owing to the restored stands providing fewer nesting-site choices and less protection from predation. Managed prairie sites were described as ecological traps by Shochat *et al*. (2005), because higher invertebrate abundances attracted breeding birds, which, subsequently, experienced poorer nesting success than in other sites. Chalfoun and Martin (2007) also documented lower nest success of Brewer's sparrow (*Spizella breweri*) in North American shrub-steppe landscapes with a greater proportion of shrub cover, despite greater densities of birds settling in these landscapes. Low-density

populations, such as those of many declining woodland bird species in Australia, face a high risk of local extinction in ecological traps (Kokko and Sutherland 2001). Many Australian woodland birds are long- lived, with a lifespan of 10–20 years being common in many species (Australian Bird and Bat Banding Scheme 2016). Consequently, there may be a time-lag before the effects of a potential ecological trap mechanism become apparent. It is, therefore, important to assess whether woodland birds are able to successfully breed in restoration plantings. In the following sections, we discuss the primary factors likely to influence the reproductive success of breeding birds in restoration plantings.

Nest predation

Predation is the primary driver of nest failure in most bird communities, causing up to 95% of failed breeding attempts (Hanski *et al*. 1996; Zanette and Jenkins 2000; Guppy *et al*. 2017; Okada *et al*. 2017). Limited work has been conducted on the effects of predation on nest success in restoration plantings internationally (Larison *et al*. 2001; Germaine and Germaine 2002), and no published studies have sought to quantify nest predation or nest success in Australian temperate woodland restoration plantings. Typical predation rates on the nests of birds vary greatly among species, even for those with similar nest structures (Ford *et al*. 2001; Weidinger 2002). For example, studies of the cup-nesting Australasian robins (Petroicidae) have consistently detected low nest success rates, in the range of 10–47%, and identified nest predation as the most common cause of failure (Robinson 1990; Zanette and Jenkins 2000; Armstrong *et al*. 2002; Debus 2006*c*). Conversely, fantails (Rhipiduridae) typically have a 59–71% nest success rate, despite building cup-nests that are less cryptic than those of robins (Cameron 1985). Parental behaviour, brood behaviour (e.g. begging), nest-site choice and concealment, and habitat variables are among several factors that may interact and contribute to highly variable nest-predation rates within and among bird communities (Martin *et al.*

2000; Haskell 2002; Weidinger 2002; Haff and Magrath 2011; Cancellieri and Murphy 2014). This variability is reflected in the diverse outcomes of nest-predation studies (e.g. Zanette and Jenkins 2000; Debus 2006*c*; Guppy *et al*. 2017), and highlights the importance of conducting such studies in restoration plantings.

Nest predation is also fundamentally dependent on the type and abundance of predators in the vicinity of the nest (Muchai and du Plessis 2005; Guppy *et al*. 2017). Avian predators cause up to 96% of nest-predation events in Australian forests and woodlands (Gardner 1998; Piper *et al*. 2002), and many predatory bird species, such as the pied currawong (*Strepera graculina*) and Australian magpie (*Cracticus tibicen*), have been favoured by habitat loss and fragmentation in temperate woodlands (Taylor and Ford 1998; Maron 2007). We might, therefore, expect to see higher rates of nest predation in restoration plantings in a fragmented landscape, where these species are more abundant, than in intact woodland remnants. Predator control may be an effective way of improving nest success in woodland birds (Debus 2006*c*), but is rarely undertaken, perhaps because of the considerable effort and resources required, in addition to the complex ecological and ethical considerations associated with controlling native predators (Wallach *et al*. 2010, 2015).

Patch size and isolation can interact with predation risk to influence breeding success and, thus, recruitment and persistence of birds in fragmented landscapes (reviewed by Stephens *et al*. 2004). Studies in fragmented landscapes worldwide have recorded lower breeding success and reproductive output in smaller habitat patches than in larger patches (Hoover *et al*. 1995; Burke and Nol 2000; Zanette and Jenkins 2000; Zanette 2001; Walk *et al*. 2010). These findings are frequently attributed to ‘edge effects’, i.e. increased nest predation near habitat edges (Hoover *et al*. 1995; Burke and Nol 2000; Willson *et al*. 2001; Vander Haegen *et al*.

2002; Herkert *et al*. 2003; Wozna *et al*. 2017). However, this notion is challenged by other studies reporting no difference in nesting success or recruitment in smaller fragments (Lehnen and Rodewald 2009; Lollback *et al*. 2010; Walk *et al*. 2010) or no evidence of edge effects increasing predator activity on nests (Hanski *et al*. 1996; Lahti 2001; Woodward *et al*. 2001; Piper *et al*. 2002; Boulton and Clarke 2003; Reino *et al*. 2010). It is important to consider the spatial scale of fragmentation relative to nest predation and its potential effects on bird populations, that is, whether fragmentation is occurring at the landscape, patch or edge scale (Zanette and Jenkins 2000; Stephens *et al*. 2004). Furthermore, different predation processes, including different primary predators, may operate in fragmented versus intact landscapes (Vander Haegen *et al*. 2002).

The contrasting outcomes of studies of nest success in fragmented landscapes imply that the effects of influential processes are either species-specific or landscape-dependent, or both. In general, we might expect species that typically experience high levels of nest predation to experience greater nest success in larger restoration plantings, or in plantings surrounded by a greater amount of vegetation cover. However, surrounding land use may have unexpected effects on the distribution and abundance of nest predators and, thus, nesting success, irrespective of patch size or connectivity. Indeed, a recent study by Okada *et al*. (2017) found effects of both nest type and the surrounding matrix (i.e. land use) on breeding success of small-bodied woodland birds in a fragmented landscape. The results were contrary to expectations; nesting success for dome-nesting species was higher in woodland patches surrounded by grazing land than in patches surrounded by pine plantations, with an abundance of avian predator nests thought to be a contributing factor. Monitoring nest predation and success is an under-utilised pathway to understanding which species are being supported in the long term, and enabling management decisions to tailor restoration programs

for species more vulnerable to predation. These topics should be thoroughly investigated in future research.

Nest-site selection

The importance of nest-site microhabitat selection in bird breeding success has been documented both internationally (Martin 1998; Mezquida 2004; Smith *et al*. 2009; Schlossberg and King 2010; Murray and Best 2014) and in Australia (Oliver *et al*. 1998; Cousin 2009; Soanes *et al*. 2015). However, research concerning woodland species nesting in restoration plantings is lacking, and may be a critical determinant of breeding success (Martin 1998). This is particularly relevant for species vulnerable to predation, such as cup-nesters (Okada *et al*. 2017). Nest-site selection for such species may act as a stronger selective pressure than other variables. For example, the western yellow robin (*Eopsaltria griseogularis*) favours sites with views of the nest surroundings over foraging opportunities when selecting a nest site (Cousin 2009), indicating that predation is a primary concern for nesting individuals of this species. It is crucial that restoration plantings provide suitable nesting-sites for a range of woodland bird species, lest they fail to support breeding populations (Larison *et al*. 2001). For example, the inclusion of trees with dense or pendulous foliage may increase availability of well-concealed nesting-sites for foliage-nesters such as the weebill and yellow thornbill. Species that nest in lower strata, such as the superb fairy-wren and speckled warbler, may be better supported with the presence of native grasses and the accumulation of dead woody material and leaf litter in the ground layer. These are factors rarely considered when constructing or monitoring restoration plantings.

Resource availability

Resource distribution and abundance in habitat patches are critical determinants of woodland bird site-occupancy and foraging patterns (Gilmore 1986; Barrett *et al*. 2008; Vesk *et al*. 2008; Montague-Drake *et al*. 2009; Munro *et al*. 2011). For example, litter and bare ground are important habitat features supporting ground-foraging birds such as robins and thornbills (Bromham *et al*. 1999; Antos and Bennett 2006). Species in these groups also prefer a low density of shrubs, as does the diamond firetail (Antos *et al*. 2008). Other species may rely on various other resources, such as woody debris; reintroduced brown treecreepers in a vegetation reserve responded positively only when woody debris was included as a habitat feature (Bennett *et al*. 2013). A lack of woody debris may be one reason the brown treecreeper is currently under-represented in restoration plantings (Martin *et al*. 2004, 2011; Lindenmayer *et al*. 2012; Gould and Mackey 2015). Furthermore, woodland bird species, including the brown treecreeper and southern whiteface, are known to vary their foraging habits and use of foraging substrates between the breeding and non-breeding seasons (Antos and Bennett 2006). This highlights the importance of using prior knowledge of species' habitat requirements to inform predicted responses of birds to habitat restoration (Bennett *et al*. 2013).

Food is generally considered a limiting resource for breeding birds (von Brömssen and Jansson 1980; Hochachka and Boag 1987; Simons and Martin 1990; Verhulst 1994; Granbom and Smith 2006; Wellicome *et al*. 2013). However, the addition of food resources does not tend to prevent major declines in fluctuating populations of terrestrial vertebrates (Boutin 1990), suggesting that the mechanisms of species decline are not usually related to resource limitation alone. Nonetheless, it is vital to assess the role of food resources in woodland bird habitat suitability. The study by Zanette *et al*. (2000) is unique in its exploration of food

shortage affecting birds in fragmented Australian woodlands; the authors documented lower availability of food resources in smaller versus larger fragments, with breeding success found to be lower in smaller fragments. Restoration plantings overwhelmingly comprise small habitat patches (Freudenberger *et al*. 2004; Smith 2008), and are known to attract a variety of bird species, including species of conservation concern (Lindenmayer *et al*. 2010*b*). When colonising sites, birds are motivated by habitat cues indicative of high resource availability, such as vegetation structure (Kokko and Sutherland 2001). If resource availability in restoration plantings does not accurately reflect these cues, then there is an increased likelihood of ecological trap mechanisms operating in revegetated landscapes (Schlaepfer *et al*. 2002).

Home range sizes of birds are inversely related to resource density and resource renewal rates (Ford 1983). This means that larger home ranges are required in habitats with fewer available resources. In a fragmented landscape, birds that are unwilling to cross habitat gaps may be disadvantaged if they are unable to expand their home ranges to exploit resources in adjacent patches (Fahrig 2007; Robertson and Radford 2009). Patchily distributed or scarce food resources can lead to inefficient foraging patterns, with subsequent reduced fitness and reproductive output in birds (Pyke 1984; Martin 1987; Granbom and Smith 2006; Flockhart *et al*. 2016). In the breeding season, optimal central-place foraging (i.e. the need to regularly return to the nest) influences searching movements, distance travelled and prey selection (Pyke 1984). In a fragmented landscape, the need to expand foraging areas or depart a patch because of resource depletion can measurably increase energy expenditure for breeding birds, thus reducing their reproductive fitness. For example, birds in fragmented landscapes may spend up to 64% more energy per chick raised than those breeding in intact remnant woodland (Hinsley *et al*. 2008). Small woodland patches have also been associated with the

contraction of breeding seasons, eggs of lighter mass being laid, and smaller nestlings being produced (Zanette *et al*. 2000). These issues could influence the breeding success of birds in restoration plantings.

For insectivorous birds in particular, dietary composition and, hence, dietary quality is directly related to habitat quality (Razeng and Watson 2012). Terrestrial invertebrates can display strong responses to habitat variables in fragmented temperate woodlands (Bromham *et al*. 1999; Barton *et al*. 2009; Lindsay and Cunningham 2009; Gibb and Cunningham 2010). As an example, Zanette *et al*. (2000) identified a 50% lower biomass of surface-dwelling invertebrates in small (55 ha) relative to large (>400 ha) woodland fragments, thereby linking food resources for insectivorous birds to patch size. Species of Coleoptera constitute the largest proportion of prey items for declining insectivorous woodland birds, followed by those of Formicidae and Lepidoptera (Razeng and Watson 2012). Coleoptera and other preferred prey of insectivorous birds have been shown to respond positively to some restoration treatments (e.g. removal of grazing pressure, addition of fallen logs to habitat patches) (Lindsay and Cunningham 2009; Gibb and Cunningham 2010). However, there is also evidence that restoration plantings may not help restore invertebrate communities in agricultural landscapes (Jellinek *et al*. 2013). It is important to understand and consider the effects of habitat fragmentation and restoration on invertebrate prey of woodland birds when assessing habitat quality in restoration plantings.

Competition

Interspecific competition for resources is a strong selective process that is enhanced in habitats with depleted or patchy resources (Cody 1981). Sought-after resources such as food and nesting sites are defended by birds in established territories, especially during the

breeding season (Robinson 1989; Broughton *et al*. 2012; Belder 2013). Closely related species may compete for similar resources, particularly food. For example, Robinson (1990) found that flame robins and scarlet robins compete more for food resources than nest sites. The noisy miner (*Manorina melanocephala*) is a strong competitor for territories and resources in Australian temperate woodlands, and actively disrupts and excludes other small woodland birds (Grey *et al*. 1998; Maron 2007; Montague-Drake *et al*. 2011; Maron *et al*. 2013; Bennett *et al*. 2015). Competition from the noisy miner has been shown to decrease breeding activity in species of smaller body mass, and can have a greater influence on woodland bird distribution and recruitment than do vegetation characteristics (Bennett *et al*. 2015; Mortelliti *et al*. 2016). Recent research has shown that the noisy miner is both increasing the risk of woodland birds going extinct from habitat patches, and decreasing the chances of them colonising patches (Mortelliti *et al*. 2016). The composition of restoration plantings can significantly affect the likelihood of colonisation and occupancy by the noisy miner; inclusion of a *Eucalyptus* overstorey increases the likelihood of noisy miner colonisation as the vegetation matures (Maron 2007). Conversely, the inclusion of an *Acacia* understorey reduces noisy miner occupancy (Lindenmayer *et al*. 2010*b*). Monitoring restoration plantings for factors likely to increase competition and competitive exclusion will provide a better understanding of species persistence mechanisms in these environments.

Brood parasitism

The influence of brood parasitism on nest success is a factor often discussed in international studies of habitat restoration (Delphey and Dinsmore 1993; Fletcher *et al*. 2006; Small *et al*. 2007; Forrester 2015), but limited research has been conducted on this topic in Australian temperate woodland ecosystems (Ford 2011; but see Guppy *et al*. 2017). There is evidence suggesting that parasitic cuckoos are dependent on large woodland remnants with an

abundance of their preferred host species, and that host species may experience greater breeding success in smaller fragments where cuckoos are rare (Brooker and Brooker 2003). Restoration plantings typically create small habitat patches (Freudenberger *et al*. 2004; Smith 2008); thus, brood parasitism events may be infrequent in revegetated sites. However, to our knowledge, no empirical studies have documented brood parasitism in temperate woodland restoration plantings, so its potential effect on the reproductive success of woodland birds in revegetated landscapes remains unknown.

Summary and future research directions

Research has shown that the responses of woodland birds to revegetation are varied, and although the habitat requirements of some species may be met, there is still much to learn about the long-term responses of birds to landscape-scale habitat restoration. Ostensibly, occupancy data alone may not expose underlying trends in population processes, or drivers of breeding success and site fidelity. To prevent and reverse the ongoing decline of Australia's woodland avifauna, and re-establish endangered habitat in highly fragmented agricultural landscapes, it is vital that temperate woodland restoration efforts continue and increase over the coming years. However, to ensure that restoration plantings are both an ecologically effective and cost- effective biodiversity conservation strategy, it is also essential for their design and management to be informed by scientific research.

There is an increasing number of modelling studies proposing strategies for optimising landscape restoration, aiming to solve the issues of catering for multiple species and ensuring maximum cost-effectiveness in the face of limited conservation resources (Bennett and Mac Nally 2004; Holzkämper *et al*. 2006; Thomson *et al*. 2007, 2009; Westphal *et al*. 2007; Lethbridge *et al*. 2010; McBride *et al*. 2010; Huth and Possingham 2011; Polyakov *et al*.

2015; Ikin *et al*. 2016). Many of these studies have provided information to help guide future restoration efforts in Australia. However, because conservation and restoration remain low priorities for governments, almost all the proposed strategies are yet to be empirically tested. Furthermore, to the best of our knowledge, all such studies are based on pattern data. Because of the lack of knowledge on population processes in revegetated landscapes, optimisation strategies for restoration to support breeding populations of woodland birds are non- existent.

Developing a comprehensive understanding of woodland bird ecology in revegetated landscapes is fundamental to devising knowledge-based solutions to reverse species decline (Bennett and Watson 2011), and a necessary key step is to move beyond pattern data, towards quantifying population responses of birds to habitat restoration. We suggest that future research in restoration plantings should focus on the areas of interest and knowledge gaps identified by the present review (summarised in Table 1.3), with an emphasis on exploring factors at the landscape- and patch-scale that are likely to contribute to restoration plantings acting as ecological traps. In particular, on the basis of our review, we suggest that the following questions should be addressed as priorities:

- What cues do birds use to select habitat in revegetated landscapes?
- Are woodland birds resident in restoration plantings in the long term?
- Do restoration plantings have higher immigration or mortality rates than do woodland remnants?
- Is habitat quality in restoration plantings sufficient for woodland birds to breed successfully?
- Does habitat suitability for breeding birds change over time as plantings mature?
- How does the breeding success of birds in plantings compare to that of birds in remnant woodland?

- What are the primary nest predators and rates of nest failure as a result of predation?
- Do restoration plantings provide suitable nesting-sites and adequate food resources for woodland birds?
- What is the role of competitive exclusion by the noisy miner?

  What is the role of brood parasitism in restoration plantings?

Finally, a more thorough approach to monitoring restored habitats is required to determine their ability to support breeding populations of woodland birds. As Battin (2004) emphasised, '. . .we cannot afford to ignore the possibility of ecological traps or fail to take them into account in the study, management, and conservation of animal populations' (p. 1490). Crucially, the capacity to accurately evaluate the success of restoration plantings in achieving intended conservation goals underpins effective utilisation of conservation resources, as well as ecologically sound environmental management.

Table 1.3 Future research directions

Summary of past and present research on birds in fragmented agricultural landscapes and landscapes undergoing habitat restoration, with recommended future research directions.

Key area	Early work		Present focus		Future directions
	Topic	Conclusions	Topic	Conclusions	Topic
Distribution and abundance	Occupancy of restoration plantings by woodland birds (e.g. Munro *et al.* 2011; Lindenmayer *et al.* 2010)	(i) Woodland bird species, including species of conservation concern, occupy restoration plantings (ii) Restoration plantings and remnant sites support different bird communities	Role of restoration plantings as habitat for woodland birds in a landscape context (e.g. Mortelliti *et al.* 2016)	Restoration plantings may not act as habitat refuges for woodland birds, including species of conservation concern	Factors influencing habitat selection by woodland birds in fragmented agricultural landscapes
Population dynamics	Ecological traps (e.g. Battin 2004)	Importance of understanding interactions between habitat selection and habitat quality	Ecological traps and undervalued resources (e.g. Gilroy and Sutherland 2007)	Understanding factors that influence colonisation of high-quality sites can inform management decisions	Quantifying habitat quality in restoration plantings; identifying potential ecological trap mechanisms in revegetated landscapes
Resources	Food resources in woodland fragments (e.g. Zanette *et al.* 2000)	Food resource availability lower in smaller than in larger woodland fragments	Resources in restored landscapes (e.g. Le Roux *et al.* 2016)	Restoration plantings may take decades to develop habitat features of remnant sites, such as nest hollows	Resource availability (food and nesting sites) in restoration plantings
	Conservation of invertebrates in woodland remnants (e.g. Barton *et al.* 2009)	Coleoptera assemblage composition closely linked to microhabitat variables e.g. fallen logs	Invertebrate community responses to habitat restoration (e.g. Gibb and Cunningham 2010; Jellinek *et al.* 2013)	Coleoptera assemblages may show either positive or neutral responses to habitat restoration	Responses of invertebrate prey of woodland birds to restoration
Breeding success	Nesting ecology of woodland birds (e.g. Robinson 1990)	Nest failures mostly due to predation	Bird breeding success in restoration plantings (e.g. Mac Nally *et al.* 2010)	Little evidence of successful breeding in restoration plantings	Quantifying nest success in restoration plantings, identifying causes of success/failure
Species interactions	Nest predation in small patches (e.g. Zanette and Jenkins 2000; Vander Haegen *et al.* 2002)	Conflicting results; nest predation may be same in small and large fragments, or increased by edge-effects in small fragments	Role of nest predation in woodland bird species declines (e.g. Debus 2006)	Intense nest predation likely cause of decline for woodland bird species of conservation concern	Quantifying nest predation, identifying primary nest predators in restoration plantings
	Brood parasitism in North American landscapes (e.g. Larison *et al.* 2001)	Brood parasitism by brown-headed cowbirds (*Molothrus ater*) lower in restored than in remnant landscapes	Brood parasitism in Australian temperate woodlands (e.g. Brooker and Brooker 2003)	Horsfield's bronze-cuckoo (*Chalcites basalis*) may be dependent on large habitat fragments	Brood parasitism in temperate woodland restoration plantings
	Influence of noisy miner on woodland bird communities (e.g. Grey *et al.* 1998)	Noisy miner disrupts and excludes small insectivorous birds from habitat patches in fragmented landscapes	Influence of noisy miner on landscape-level bird species distribution patterns (e.g. Mortelliti *et al.* 2016)	Noisy miner main driver of bird distribution patterns in fragmented woodlands, prevents restoration plantings acting as habitat refuges	Effects of noisy miner removal on landscape-level bird species distribution patterns and restoration planting occupancy

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CHAPTER 2

Is bigger always better? Influence of patch attributes on breeding activity of birds in box-gum grassy woodland restoration plantings

Fenceline between a small restoration planting and surrounding pasture on the property “Boorook”, Morven, NSW. Photo: Donna Belder.

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Abstract

Restoration plantings are an increasingly common management technique to address habitat loss in agricultural landscapes. Native fauna, including birds, may readily occupy planted areas of vegetation. However, unless restoration plantings support breeding populations, their effectiveness as a conservation strategy may be limited. We assessed breeding activity of birds in box-gum grassy woodland restoration plantings in the South-west Slopes bioregion of New South Wales, Australia. We compared breeding activity in plantings of different size (small and large) and shape (linear and block-shaped) to breeding activity in a set of remnant woodland sites. Contrary to expectations, we found that bird breeding activity was greatest per hectare in small patches. We also found a negative effect of planting age, with younger plantings supporting more breeding activity per hectare. We found no effect of patch type or shape on breeding activity, and that species' relative abundance was not predictive of their degree of breeding activity. Our results highlight the value of small habitat patches in fragmented agricultural landscapes, and indicate that restoration plantings are as valuable as remnant woodland patches for supporting bird breeding activity. We demonstrate the importance of breeding studies for assessing the conservation value of restoration plantings and other habitat patches for avifauna.

Keywords: Woodland birds, breeding success, SLOSS, restoration, fragmentation, agricultural landscapes

Introduction

Habitat loss due to land conversion for agriculture is a significant issue globally, with numerous effects on biodiversity and ecosystem processes (Tscharntke *et al.* 2012; Maxwell *et al.* 2016). Land clearing is increasing worldwide, particularly in productive agricultural regions (Evans 2016; Tilman *et al.* 2017). The extensive removal of native vegetation creates a highly fragmented landscape in which patches of native vegetation exist primarily as small, isolated remnants. Restoration plantings in agricultural landscapes are increasingly implemented to address habitat loss and conserve threatened and declining native fauna, with hundreds of millions of hectares of vegetation being replanted around the world at a cost of billions of dollars (Crouzeilles *et al.* 2016). To ensure cost-effectiveness and ecological integrity, it is important to quantify the effects of revegetation on biodiversity and assess whether conservation goals are being met, particularly in the long term (Ruiz-Jaen and Aide 2005; Barral *et al.* 2015).

A core assumption of restoration success is that revegetated patches provide high-quality habitat for the species they are intended to help conserve (Ruiz-Jaen and Aide 2005; Ikin *et al.* 2016). In Australia, bird communities that inhabit box-gum grassy woodlands are threatened by ongoing habitat loss and degradation (Rayner *et al.* 2014), and are a frequent target of restoration efforts (Freudenberger 2001; Smith 2008; Lindenmayer *et al.* 2013). There is evidence suggesting that many bird species will readily occupy restoration plantings, in some cases preferentially inhabiting plantings over remnant woodland patches or other sites (Barrett *et al.* 2008; Cunningham *et al.* 2008; Lindenmayer *et al.* 2016), but how much do we know about the capacity of restoration plantings to support breeding populations of these species? The majority of studies examining avian responses to restoration plantings have used measures such as species richness, diversity, and relative abundance to make

inferences about occupancy trends and habitat quality (Belder *et al.* 2018). However, focusing on occurrence patterns provides a limited picture of how birds are using a site (Chalfoun and Martin 2007). It is therefore important to quantify whether indicators of long-term persistence, such as breeding activity, follow the same trends.

Research objectives

The underlying aim of this study was to assess whether birds are able to breed successfully in box-gum grassy woodland restoration plantings. Breeding success can be measured in several ways, with nest success and daily nest survival being commonly used metrics (Stephens *et al.* 2004). However, searching for, and monitoring, nests requires considerable time and effort. An alternative, and perhaps more accessible, approach is to use indicators of breeding activity as a proxy for breeding success. For example, a scoring system developed by Mac Nally (2007) ranks observations of breeding behaviour according to how strongly they indicate breeding success (Table 2.1), providing a quantitative measure of the extent to which a given site supports successful breeding (Selwood *et al.* 2009; Mac Nally *et al.* 2010; Bennett *et al.* 2015). A method such as this provides a basis from which to commence the transition from traditional occupancy and abundance surveys to a more population-oriented approach to monitoring avian responses to restoration plantings. Importantly, it also facilitates the collection of breeding data on species of conservation concern, whose nests may be difficult to find in adequate numbers.

We sought to investigate bird breeding activity in the context of habitat restoration in a fragmented agricultural landscape. Specifically, we posed the following three questions:

Question 1. How does bird breeding activity in restoration plantings compare to breeding activity in remnant woodland patches?

We compared breeding activity in restoration plantings, similar-sized woodland remnants, and larger, more intact woodland remnants. In addition to investigating the entire bird assemblage, we assessed breeding activity for species of conservation concern, and cup-nesters vs. dome nesters (Appendix 2.2). Remnant patches are generally considered to be high-value habitat within fragmented agricultural landscapes (Cunningham *et al.* 2014), and hence we predicted remnant sites would support more breeding activity than restoration plantings. We predicted that breeding activity would be highest in larger woodland remnants than in smaller, more isolated remnants and restoration plantings. We made this prediction because comparative studies have shown that species richness and abundance is typically highest in large, intact remnants (Helzer and Jelinski 1999; Martin *et al.* 2004; Munro *et al.* 2011; Hadley *et al.* 2018). Many species of conservation concern are more closely associated with remnants than plantings (Kinross 2004), so we also expected to observe more breeding activity from these species in remnants than in plantings.

Question 2. How do patch attributes affect breeding activity in plantings and remnant woodland patches?

We examined breeding activity in sites of varying size (small and large) and shape (linear and block-shaped). A key finding from pattern-based studies of bird distribution and abundance in fragmented landscapes is that larger patches support more species (Watson *et al.* 2003; Kavanagh *et al.* 2007; Shanahan *et al.* 2011). This is consistent with the resource concentration hypothesis, which posits that there are more resources and thus more individuals and greater species diversity in larger patches (Root 1973; Connor *et al.* 2007). Previous species-specific studies have also found that avian reproductive success is positively

correlated with patch size (Hoover *et al.* 1995; Zanette *et al.* 2000; Herkert *et al.* 2003; Luck 2003). We therefore postulated that breeding activity would increase with patch size in parallel with bird species richness and abundance. Similarly, increasing patch linearity is typically associated with lower species richness and abundance (Kinross 2004; Lindenmayer *et al.* 2007, 2018a). As such, we predicted more evidence of successful breeding in block-shaped than in linear patches.

We predicted a stronger negative response to decreasing patch size and increasing linearity for cup-nesters compared with dome-nesters. This was because edge-effects of predation are stronger in smaller and more linear sites (Helzer and Jelinski 1999; Fletcher *et al.* 2007), and cup-nesters tend to be more vulnerable to predation than other nest types (Okada *et al.* 2017). We also predicted that species of conservation concern, many of which are area-sensitive (Watson *et al.* 2005; Ford *et al.* 2009), would show more evidence of breeding activity in larger, block-shaped sites.

We also tested for an effect of planting age. Previous studies report increases in bird species richness and abundance as plantings mature (Freeman *et al.* 2009; Lindenmayer *et al.* 2016; Debus *et al.* 2017). This is often attributed to the tendency of the vegetation structure and composition of restoration plantings to converge on that of remnant patches over time (Munro *et al.* 2011). We therefore predicted that increasing planting age would have a positive effect on bird breeding activity.

Question 3. Does breeding activity in restoration plantings and remnant woodland patches reflect species assemblage composition?

We predicted that breeding activity in our study sites would be reflective of the species assemblage present. That is, we expected the effects of patch attributes (type, size, shape) on relative abundance to be correlated with the effects of patch attributes on breeding activity scores.

Methods

Study area

We conducted this study in the South-west Slopes bioregion of New South Wales, Australia (Figure 2.1). The region is part of Australia's sheep-wheat belt and has been extensively cleared of native vegetation, with as little as 0.1% of the original vegetation remaining in intact condition (Thiele and Prober 2000). Remnant patches consist predominantly of white box (*Eucalyptus albens*) / yellow box (*E. melliodora*) / Blakely's red gum (*E. blakelyi*) grassy woodland, which is a critically endangered ecological community (Department of the Environment 2018). Patches of red stringybark (*E. macrorhyncha*) woodland and mugga ironbark (*E. sideroxylon*) woodland are also present in our study region.

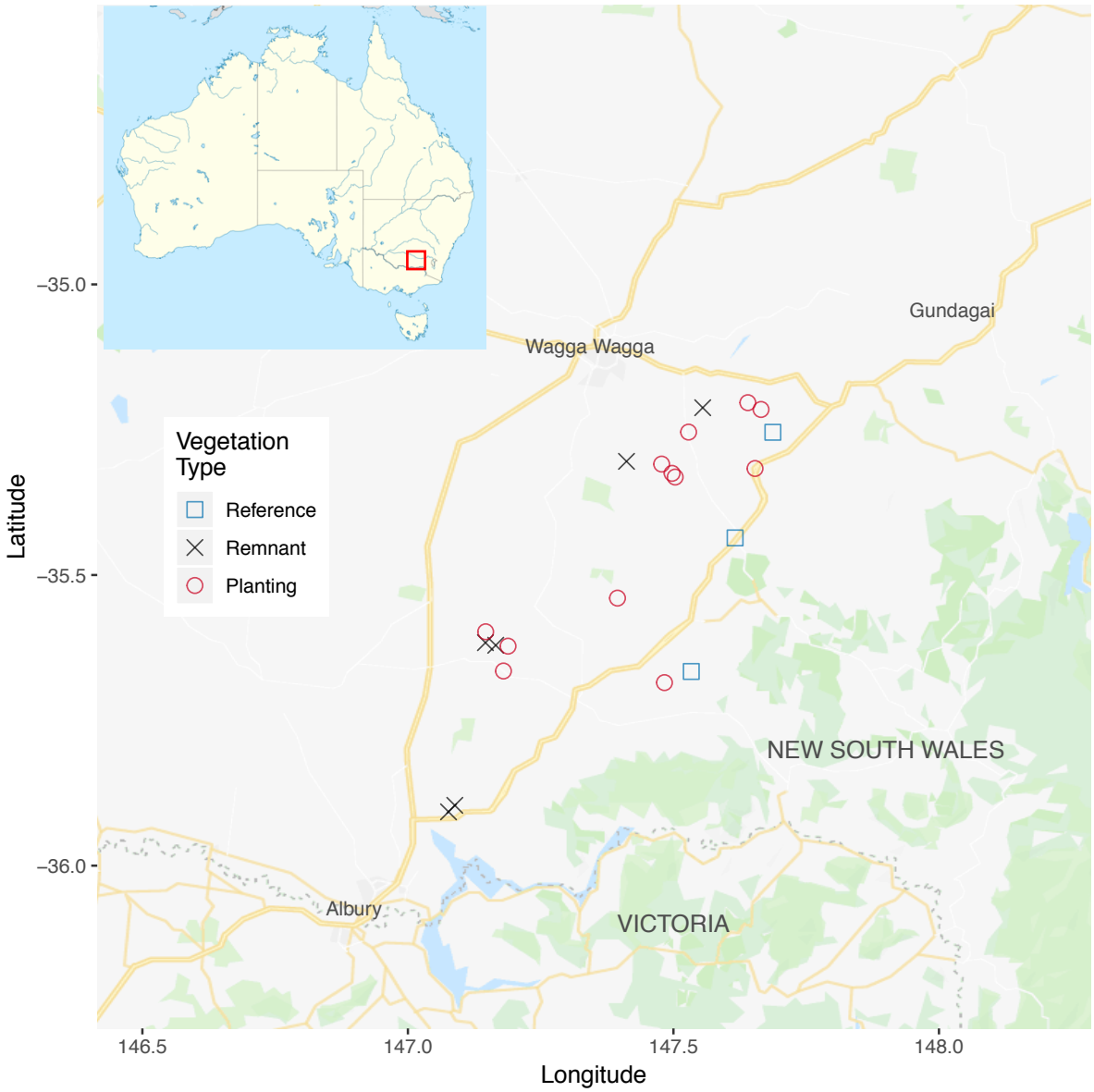

Figure 2.1 Location of study sites in the South-west Slopes Bioregion of New South Wales, Australia. Map created using ggmap for R (Kahle and Wickham 2013).

Study sites

We used spring bird survey data collected over 12 years to select a subset of 12 restoration plantings from a set of long-term monitoring sites (Appendix 2.1) (Cunningham *et al.* 2007). We selected sites on the basis that they satisfied our criteria for size and shape, and shared at least two of three key species in common – the superb fairywren (*Malurus cyaneus*), yellow-rumped thornbill (*Acanthiza chrysorrhoa*), and willie wagtail (*Rhipidura leucophrys*). We chose these species as they are relatively common, typically found in woodland communities,

and encompass the two major nest types (one cup-nester and two dome-nesters). Additionally, the yellow-rumped thornbill is a species of conservation concern (Barrett *et al.* 2003). Nineteen of our 21 chosen sites contained all three target species, with two sites lacking the yellow-rumped thornbill. We attempted to control for the effects of competitive exclusion by selecting sites with low abundances of the noisy miner (*Manorina melanocephala*), as this hyper-aggressive species is known to have negative impacts on other species of native birds (Maron *et al.* 2013; Bennett *et al.* 2015). Our sites were separated geographically by at least 500 m to promote spatial independence.

Plantings were aged between 12 and 25 years, 1.3-7.7 ha in area, and 20-200 m in width. A typical planting contained a mature (flowering-age) *Eucalyptus* overstorey, an *Acacia* understorey, and a ground layer dominated by annual grasses (both native and exotic). The majority of planted species naturally occur in the study region. Some plantings also contained remnant trees, along with varying amounts of woody debris (fallen trees and branches).

We compared plantings with six box-gum grassy woodland remnants, also part of the long-term monitoring study. Remnant patch size ranged from 2.1 to 5.8 ha, with widths of 30-200 m. We also selected three large (47-110 ha) reference sites to represent intact remnant woodland in the study region (two travelling stock reserves, and one remnant on private property). Remnant sites were dominated by a *Eucalyptus* overstorey, with or without an *Acacia* understorey, and typically contained woody debris in the form of fallen trees and branches.

Bird surveys

To assess breeding activity, we conducted fixed time-per-unit-area surveys (one hour per hectare) in our study sites over two spring breeding seasons. The peak breeding season for the majority of bird species in our study region is September to December (Appendix 2.2). We completed two rounds of surveys in 2015 (October and November), and three rounds in 2016 (September, October, November). We searched sites systematically, identifying and recording indicators of breeding behaviour (Table 2.1). We designated search areas by the size and shape of sites. For sites < 3 ha, we searched 1.3 ha within the site – this was equivalent to the area of the smallest study site. For sites > 3 ha, we searched 3 ha within the site. We surveyed block sites in a grid fashion, and linear sites along their length until we had searched the desired area (i.e. 1.3 ha or 3 ha). We surveyed sites throughout the day, with the exception of November 2016 – in this period we completed surveys in the 4 hours post-sunrise and 4 hours pre-sunset. On average, there was an interval of 4.5 weeks between surveys at each site, and we structured the order of site visits to ensure that sites were not consistently surveyed at the same time of day. We did not conduct surveys during inclement weather. All breeding activity surveys were conducted by Author 1.

Table 2.1 Scores allocated to behavioural observations of breeding activity, modified from Mac Nally (2007).

Behaviour	Score
Feeding of young out of the nest	9.0
Fledglings seen	9.0
Nest with nestlings or feeding of young in the nest	8.0
Presence of juveniles or immature birds	7.5
Fledglings heard	7.5
Adult carrying food	6.0
Nest with eggs or adult on a nest	6.0
Nest empty or under construction (current breeding season)	5.0
Past breeding season's nest	3.5
Adult gathering nest material	3.0
Courtship	2.0
Territorial behaviour	1.0
Male and female pairs	1.0

To quantify breeding activity, we used a survey method modified from Mac Nally's (2007) scoring system. The Mac Nally (2007) method involves calculating an aggregate score of breeding activity in a study site over the course of a study. Scores are calculated based on ranking observations according to how strongly they indicate breeding success (Table 2.1), with a score of zero indicating no observations of breeding activity. Rather than aggregating breeding activity scores over the course of the study, we modified the method to calculate a score per survey. There were two reasons for this: first, it enabled us to test for effects of factors that may influence detectability of bird behaviour during surveys, such as weather and time of day. Second, it enabled us to account for repeat observations of the same individuals or nests across multiple surveys.

We conducted point count surveys in 2016 to quantify bird community composition and abundance in our study sites. Point count surveys in each site were typically conducted within two days of the surveys for breeding activity, and usually on the same day. Point count surveys in September were conducted by Author 1, and in October and November were

completed by different observers (the entirety of each month's surveys conducted by a different observer). We divided each study site into 25 x 25 m cells, and randomly selected cells in which to conduct point counts. For sites > 3 ha, we selected six cells, and for sites <3 ha, we selected three cells. We ensured adjacent cells were not selected. At the centre of each randomly chosen cell, we completed a five-minute count, recording counts of birds detected within 50 m of the survey point.

Statistical analyses

We used a model selection approach to investigate the effects of patch attributes on the total breeding activity score recorded in each survey (Table 2.2). We used linear mixed effects regression models with study site and survey year as random effects to account for repeated visits to sites over multiple years. The explanatory variables of primary interest were site type, size, and shape, and age of plantings. We included the variable "fenced", to account for potential effects of cattle grazing in our study sites (Lindenmayer *et al.* 2018*b*). Our response variable was a total breeding activity score standardised by survey area (1.3 or 3.0 ha), and was square-root-transformed to improve the distribution of the data. We also scaled and centred our continuous predictor variables. Prior to fitting models with our explanatory variables of interest, we examined variables likely to influence detectability in surveys, including time of day, temperature, and wind. In addition, we accounted for variation in activity through the breeding season by including Julian date. We found that breeding activity increased with Julian date for the woodland assemblage and all subsets of the assemblage, so included it as an explanatory variable in subsequent models. There were no other weather or temporal variables of statistical significance (Appendix 2.4).

Prior to fitting models, we checked all explanatory variables for multi-collinearity using variance inflation factors. We corrected for multi-collinearity by removing large reference sites from models that included both size and shape. We also removed temperature due to its correlation (0.53) with time of day. We checked for a quadratic effect of time of day and found none. After fitting models, we checked for spatial autocorrelation in the data using variograms of the residuals. We detected no evidence of a nugget or sill in the variograms, and therefore assumed no spatial autocorrelation.

For our analyses, we included data for all terrestrial species recorded during breeding activity surveys, with the exception of introduced species (Appendix 2.2). We hereafter refer to this assemblage as the "woodland assemblage". For babblers and finches, we included data on nests only when they could be positively identified as true nests – these species build roost nests, which can be difficult to distinguish from true nests. We subset the woodland assemblage to investigate species of conservation concern, and compared cup-nesters with dome-nesters. We defined species of conservation concern as those listed as threatened in New South Wales (NSW Environment and Heritage 2018), along with those whose reporting rates declined by >20% in the South-west Slopes bioregion between the first and second *Atlas of Australian Birds* (Barrett *et al.* 2003). We classified cup-nesters and dome-nesters as per Morcombe (2003) and Pizzey and Knight (1997). The dome-nester group was highly correlated (0.79) with the woodland assemblage, as were species of least concern (0.91), so we did not analyse these groups separately. In addition to examining species of conservation concern and cup-nesters, we subset the woodland assemblage data to remove the most dominant species (superb fairywren, yellow-rumped thornbill, and willie wagtail).

For the woodland assemblage, and each subset, we followed a three-step modelling approach:

1. We first accounted for variation in our response variable associated with weather and temporal factors. We incorporated variables of significance into subsequent models.
2. We then modelled our response variable against site type, comparing plantings, remnants, and large reference sites.
3. Finally, we modelled our response variable against size and shape in plantings and remnants, excluding large reference sites.

In each step, we fitted global models with all combinations of the variables of interest, and ranked candidate models using Akaike's Information Criterion corrected for small sample sizes (AIC_c). We considered models with $\Delta AIC_c \leq 2$ as top-ranked models (Burnham and Anderson 2004). Weather and temporal variables of significance identified in Step 1 were included in both Step 2 and Step 3.

We used the packages 'lme4' (Bates *et al.* 2015) and 'MuMIn' (Bartoń 2018) in R version 3.4.4 (R Core Team 2018) to fit and select models. Variograms were constructed using the package 'geoR' (Ribeiro and Diggle 2016).

Table 2.2 Linear mixed model parameters. The response variable is SCORE, and all other variables are predictors.

Variable name	Description
SCORE	Square root of score of breeding activity recorded during surveys, calculated per Mac Nally (2007) and standardised by survey area (score/1.3 for small sites, score/3.0 for large sites)
TYPE	Site type (planting, remnant, reference)
SIZE	Site size (ha)
SHAPE	Measure of site shape, calculated as perimeter/width (m)
AGE	Age of planting at the commencement of the study (years)
FENCED	Site fenced from cattle (yes/no)
SUN	Subjective measure of sun during surveys, on a numerical scale of 1-4 where 1 = full sun and 4 = overcast
TEMP	Subjective measure of temperature during surveys, on a numerical scale of 1-8 where 1 = cold and 8 = hot
WIND	Subjective measure of wind during surveys, on a numerical scale of 1-8 where 1 = calm and 8 = strong wind
TIME	Time of day surveys commenced, given as no. hours post-sunrise (hr)
DATE	Julian date on which surveys were conducted

We used multivariate latent variable models from the package 'boral' (Hui 2016) to compare how abundance and breeding activity for bird species responded to site type, size, and shape. This approach is useful because it allows for investigation of the association between multiple species and underlying environmental variables in a linear modelling framework, while also accounting for potential correlations among species. Specifically, we constructed one latent variable model for each response matrix, and then compared the coefficient estimates for each species and variable. For this modelling approach, only species detected both in point count surveys and breeding activity surveys could be included. We subset our data to an assemblage of interest that included woodland-dependent species (Silcocks *et al.* 2005) and several other small-bodied species that characterise the bird community of woodlands in our study region (Appendix 2.3). Due to the disproportionate spatial influence of the frequently detected superb fairywren in our initial ordination plots, we excluded it from our multivariate latent variable models.

Results

General findings

A total of 90 bird species was detected during point count surveys, of which 66, or 73%, displayed evidence of breeding activity (Appendix 2.2). Additionally, two species – the hooded robin (*Melanodryas cucullata*) and brown goshawk (*Accipiter fasciatus*) – were recorded in breeding activity surveys but not detected in point counts. The most commonly detected species was the superb fairywren, which accounted for 26% of all breeding activity recorded in the study. Other frequently detected species were the willie wagtail, yellow-rumped thornbill, grey shrikethrush (*Colluricincla harmonica*), and rufous whistler (*Pachycephala rufiventris*). The species of conservation concern we detected during surveys included the yellow-rumped thornbill, weebill (*Smicrornis brevirostris*), speckled warbler (*Pyrrholaemus sagittatus*), dusky woodswallow (*Artamus cyanopterus*), crested shrike-tit (*Falcunculus frontatus*), and hooded robin. For the woodland assemblage, breeding activity scores recorded during surveys ranged from 11.5 to 104.5, with a mean of 46.0 (n=105, SE=2.2). The mean score for cup-nesters was 19.1 (n=105, SE=1.4), with minimum and maximum scores of 0 and 76.0 respectively. For species of conservation concern, the mean score was 11.5 (n=105, SE=1.4), minimum score 0, and maximum score 55.0. We found no differences in breeding activity in sites that were fenced compared with sites that were exposed to grazing by stock.

How does woodland bird breeding activity in restoration plantings compare to breeding activity in remnant woodland patches?

For the woodland assemblage, the score for breeding activity did not differ between plantings, remnants, and reference sites (Appendix 2.5). That is, site type did not appear as a variable of significance in any of our top-ranked models. The same was true when comparing only

plantings and remnants (excluding reference sites) (Table 2.3). We found no effect of site type on species of conservation concern, and cup-nesters showed no response to site type. Removing the superb fairywren, willie wagtail and yellow-rumped thornbill from the woodland assemblage did not elicit any response to site type from the remainder of the assemblage.

How do patch attributes affect breeding activity in plantings and remnant woodland patches? Modelling patch attributes of remnants and plantings (excluding large reference sites) against breeding activity score for the woodland assemblage revealed a strong negative effect of increasing patch size, which appeared consistently in the top two candidate models (Table 2.3). That is, there was more breeding activity per hectare in smaller patches (Figure 2.2). However, the removal of the superb fairywren from the woodland assemblage greatly reduced the negative effect of site size on breeding activity (Table 2.3). Size appeared as an explanatory variable in candidate models for breeding activity score of assemblages without superb fairywren, willie wagtail, and yellow-rumped thornbill, but its inclusion did not substantially improve the fit of the simplest model (containing only Julian date). Where size appeared as an explanatory variable, its effect was marginal, with a large standard error.

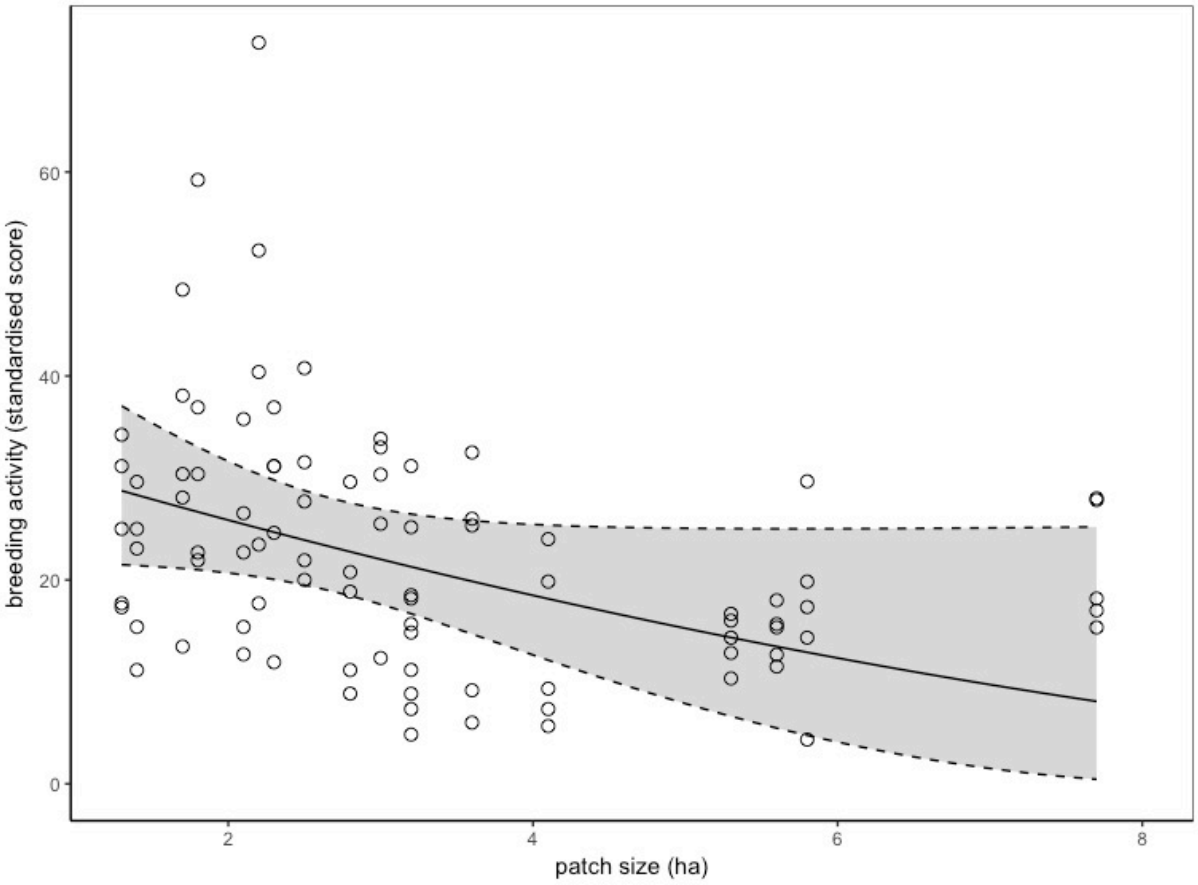

Figure 2.2 Effect plot illustrating the influence of patch size on breeding activity score of the woodland assemblage in restoration plantings and similarly sized woodland remnants. Shading indicates 95% confidence intervals. Plot created using ggplot2 for R (Wickham 2016).

Excluding large reference sites revealed a marginal negative effect of site type, suggesting that breeding activity score was higher in plantings than in similarly sized woodland remnants (Table 2.3). However, the inclusion of site type did not substantially improve the fit of the simplest model, and it failed to appear in top-ranked models after the removal of the three dominant species from the assemblage. Shape appeared in one top-ranked model when the superb fairywren was excluded from the assemblage, however, the standard error was larger than the effect size itself. Site shape therefore had no interpretable effect on breeding activity score.

For species of conservation concern, the best fitting model was the model containing only Julian date (Table 2.3). Consequently, there were no interpretable effects of planting size, shape, or type on breeding activity score for this subset. The same result was observed for cup-nesters, again indicating marginal or no effects of patch attributes on breeding activity score. Dome-nesters mirror the negative response to patch size demonstrated by the woodland assemblage (per 0.81 correlation).

Table 2.3 Parameter estimates for total breeding score recorded during breeding activity surveys, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown.

Woodland assemblage	Rank 1 (w = 0.22)	Rank 2 (w = 0.12)	
	Estimate (SE)	Estimate (SE)	
Intercept	4.57 (0.22)	4.69 (0.24)	
DATE	0.59 (0.09)	0.58 (0.09)	
SIZE	– 0.40 (0.16)	– 0.39 (0.16)	
TYPE (remnant)		– 0.37 (0.33)	
Excluding superb fairywren	Rank 1 (w = 0.19)	Rank 2 (w = 0.11)	Rank 3 (w = 0.08)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	3.83 (0.12)	3.83 (0.12)	3.69 (0.12)
DATE	0.38 (0.10)	0.38 (0.10)	0.37 (0.10)
SIZE		– 0.13 (0.12)	– 0.53 (0.21)
SHAPE			– 0.06 (0.11)
SIZE:SHAPE			– 0.67 (0.32)
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	Rank 1 (w = 0.20)	Rank 2 (w = 0.09)	Rank 3 (w = 0.09)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	3.19 (0.15)	3.19 (0.14)	3.38 (0.27)
DATE	0.24 (0.10)	0.24 (0.10)	0.24 (0.10)
SIZE		– 0.12 (0.14)	
FENCED (yes)			– 0.26 (0.32)

Table 2.3 cont.

Species of conservation concern	Rank 1 (*w* = 0.13)	Rank 2 (*w* = 0.11)	Rank 3 (*w* = 0.07)	Rank 4 (*w* = 0.06)	Rank 5 (*w* = 0.06)	Rank 6 (*w* = 0.05)
	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	1.76 (0.23)	1.99 (0.27)	1.38 (0.43)	1.76 (0.23)	1.99 (0.26)	2.00 (0.25)
DATE	0.25 (0.10)	0.24 (0.10)	0.24 (0.10)	0.25 (0.10)	0.23 (0.10)	0.24 (0.10)
TYPE (remnant)		– 0.69 (0.47)			– 0.71 (0.46)	– 0.67 (0.43)
FENCED (yes)			0.52 (0.51)			
SIZE				0.21 (0.23)	0.23 (0.22)	0.37 (0.23)
SIZE:TYPE (remnant)						– 0.77 (0.53)
Cup-nesters	Rank 1 (*w* = 0.23)	Rank 2 (*w* = 0.10)	Rank 3 (*w* = 0.09)			
	Estimate (SE)	Estimate (SE)	Estimate (SE)			
Intercept	2.58 (0.21)	2.47 (0.25)	2.58 (0.20)			
DATE	0.27 (0.10)	0.27 (0.10)	0.27 (0.10)			
TYPE (remnant)		0.32 (0.43)				
SHAPE			0.14 (0.21)			

Planting age was a significant predictor of breeding activity for the woodland assemblage (Table 2.4). An increase in planting age was associated with a decrease in breeding activity. This result was no longer evident when the superb fairywren, yellow-rumped thornbill and willie wagtail were removed from the assemblage. However, species of conservation concern also responded negatively to an increase in planting age. No effect of planting age was observed for cup-nesters. For the latter subset, the null model was the top-ranked model.

Table 2.4 Parameter estimates for total breeding score recorded during breeding activity surveys, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown. Note that candidate models for cup-nesters are not included, as the null model was the top-ranked model for this subset.

Woodland assemblage	Rank 1 (w = 0.27)	Rank 2 (w = 0.22)	Rank 3 (w = 0.10)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	5.65 (0.51)	4.73 (0.25)	4.73 (0.25)
DATE	0.52 (0.11)	0.54 (0.11)	0.54 (0.11)
AGE	– 0.42 (0.15)	– 0.51 (0.18)	– 0.51 (0.18)
SIZE	– 0.37 (0.14)	– 0.41 (0.18)	– 0.45 (0.19)
FENCED	– 0.99 (0.53)		
SHAPE			– 0.17 (0.19)
Excluding superb fairywren	Rank 1 (w = 0.20)	Rank 2 (w = 0.13)	Rank 3 (w = 0.13)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	3.85 (0.12)	3.85 (0.14)	3.30 (0.42)
DATE	0.32 (0.12)	0.31 (0.12)	0.32 (0.12)
AGE	– 0.25 (0.13)		– 0.30 (0.12)
FENCED			0.60 (0.45)
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	Rank 1 (w = 0.22)	Rank 2 (w = 0.12)	
	Estimate (SE)	Estimate (SE)	
Intercept	3.16 (0.19)	3.16 (0.19)	
DATE	0.21 (0.11)		
Species of conservation concern	Rank 1 (w = 0.20)	Rank 2 (w = 0.12)	
	Estimate (SE)	Estimate (SE)	
Intercept	1.99 (0.21)	1.99 (0.26)	
DATE	0.30 (0.13)	0.29 (0.13)	
AGE	– 0.53 (0.21)	– 0.52 (0.27)	
SIZE	0.43 (0.21)		

Does breeding activity in restoration plantings and remnant woodland patches reflect species assemblage composition?

Based on ordination modelling, we found that breeding activity was not strongly correlated with relative abundance for bird species in our study sites (Figure 2.3). Examining the effects of patch attributes on relative abundance and breeding activity revealed that many species differed in their responses according to the two metrics. For example, the abundance of the willie wagtail in remnants and plantings was similar, but breeding activity for this species was higher in remnants (Figure 2.3a). A similar pattern was observed for the buff-rumped thornbill (*Acanthiza reguloides*), for which more breeding activity was recorded in large reference sites than in plantings, despite the species occurring in similar abundances in the two site types (Figure 2.3b). The buff-rumped thornbill also displayed a positive response to increasing patch size according to relative abundance, but a negative response according to breeding activity (Figure 2.3c). Interestingly, there were no species whose abundance increased with patch linearity, but several species, including the black-faced cuckooshrike (*Coracina novaehollandiae*), brown treecreeper (*Climacteris picumnus*), and willie wagtail, showed more evidence of breeding activity in linear sites (Figure 2.3d). We note that confidence intervals around the estimates for many species were large (Appendix 2.8).

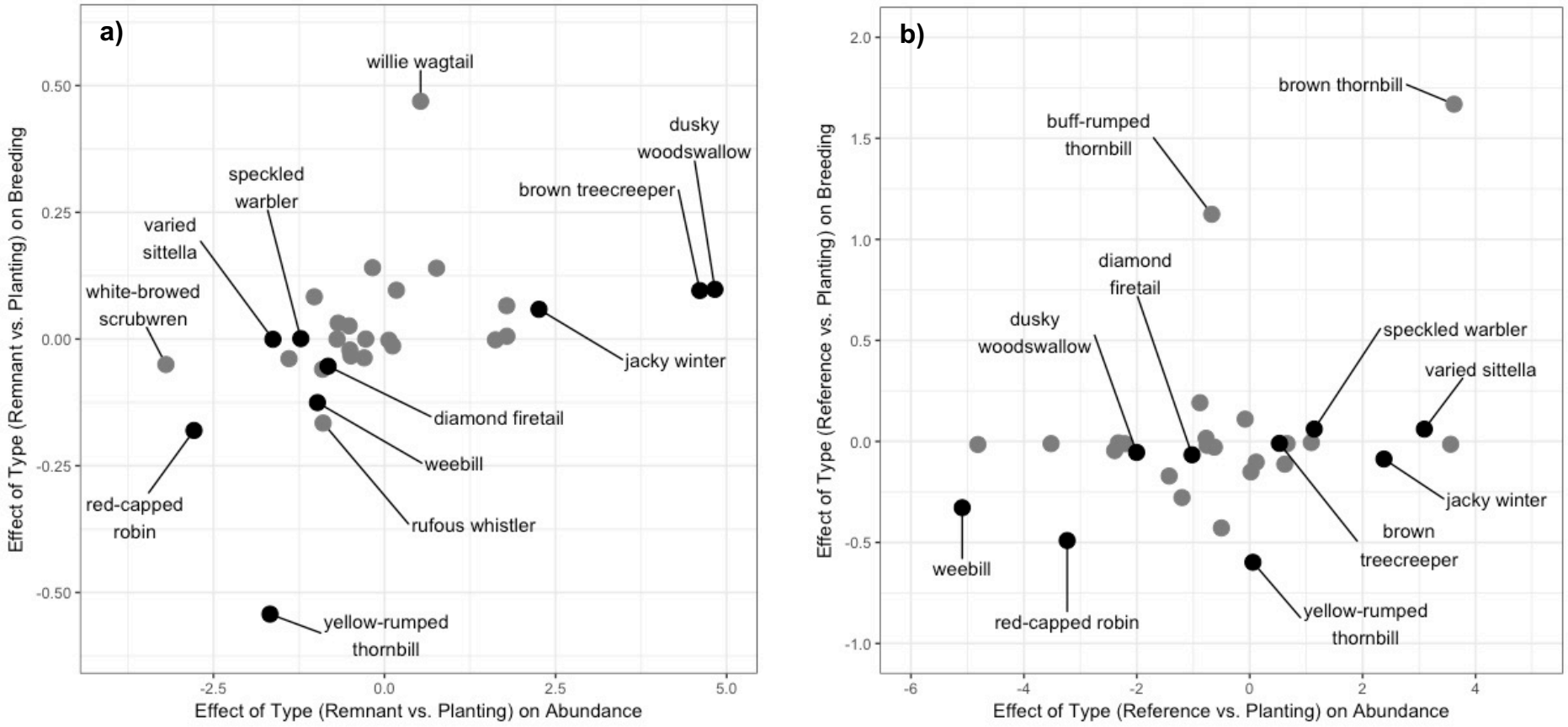

Figure 2.3 Bird species' relative abundance and breeding activity plotted according to the effects of a) patch type: remnants vs. plantings, in which a positive effect is associated with remnants, b) patch type: reference sites vs. plantings, in which a positive effect is associated with reference sites, c) patch size, and d) patch shape, in which the effect becomes more negative with increasing patch linearity. Effect sizes are taken from multivariate latent variable models. ● = species of conservation concern, ● = species of least concern. Plots created using ggplot2 (Wickham 2016) and ggrepel (Slowikowski 2018).

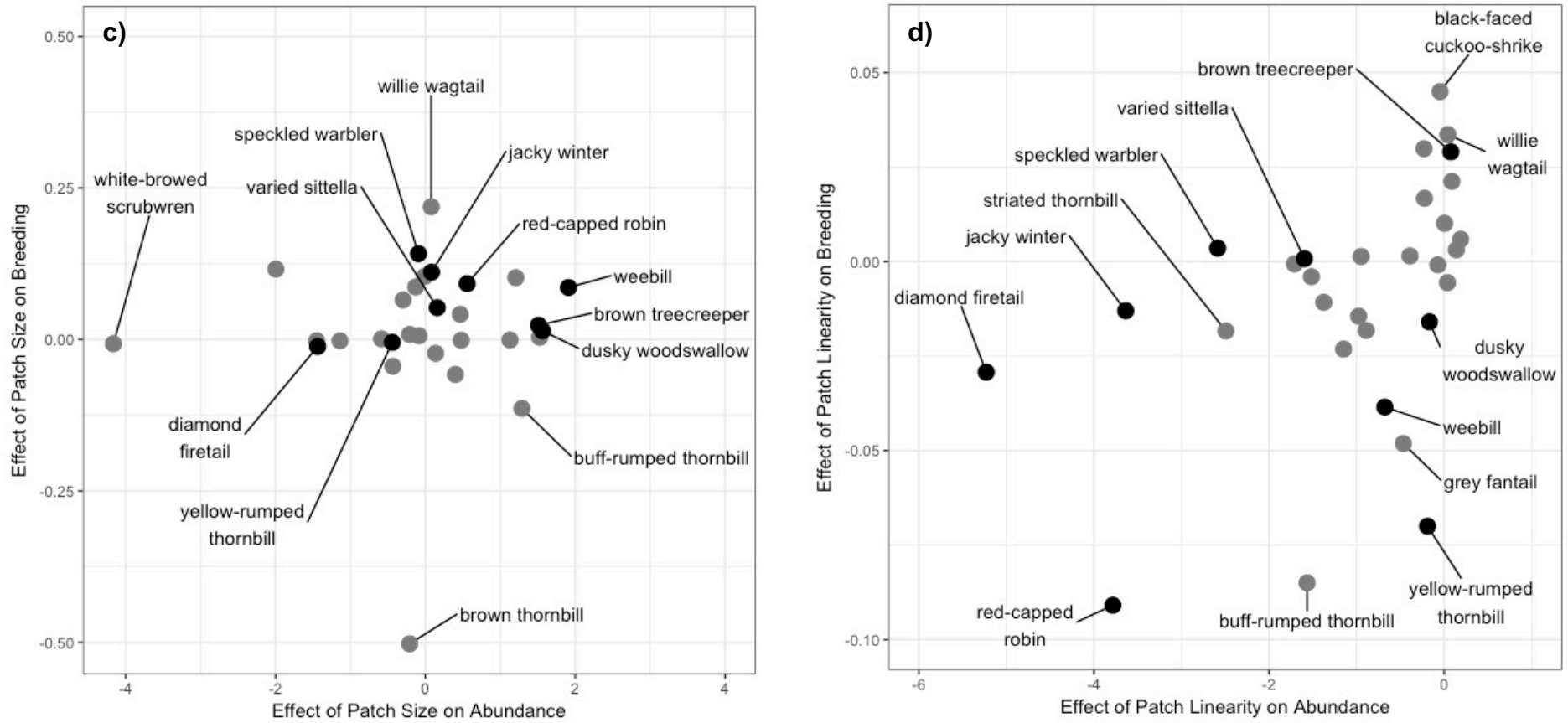

Figure 2.3 Bird species' relative abundance and breeding activity plotted according to the effects of a) patch type: remnants vs. plantings, in which a positive effect is associated with remnants, b) patch type: reference sites vs. plantings, in which a positive effect is associated with reference sites, c) patch size, and d) patch shape, in which the effect becomes more negative with increasing patch linearity. Effect sizes are taken from multivariate latent variable models. ● = species of conservation concern, ● = species of least concern. Plots created using ggplot2 (Wickham 2016) and ggrepel (Slowikowski 2018)

Discussion

We recorded breeding activity of a variety of bird species in both restoration plantings and remnant woodland patches. Our analyses of the effects of patch attributes revealed several unexpected findings – most notably, a negative effect of patch size driven by one dominant species (the superb fairywren), in which there was more breeding activity per hectare in smaller patches. Inferences from our study contrast with those of numerous studies on bird species richness and abundance in fragmented agricultural landscapes, which report a positive effect of patch size. We further discuss our key findings in the remainder of this paper and conclude with some insights for bird conservation.

Patch size

Contrary to our predictions at the outset of this study based on patch size theory (Rosenzweig 1995), we found that breeding activity score per hectare decreased as patch size increased in plantings and remnant woodland patches. This result was driven by the most commonly detected species in the study region, and when this species was removed, there was no effect of patch size on breeding activity. Both of these findings contrast with the majority of previous studies, which have documented higher breeding success and reproductive output in larger habitat patches than in smaller patches (e.g. Burke and Nol 2000; Zanette and Jenkins 2000; Zanette *et al.* 2000; Zanette 2001). The value of small habitat patches for biodiversity in fragmented landscapes has been highlighted via studies of bird species distribution and abundance (Fischer and Lindenmayer 2002; Gibbons and Boak 2002; Manning *et al.* 2006; Le Roux *et al.* 2015), and was underscored by Wintle *et al.* (2019) in their global synthesis of conservation studies. Our results indicate that small patches may play a substantial role in supporting bird populations, which we discuss further in the concluding sections of this paper.

According to island biogeography theory, which has been applied to fragmented agricultural landscapes, smaller patches may stimulate a concentration effect of animal populations in fragmented landscapes (MacArthur and Wilson 1967). For example, waterbirds have been recorded breeding in greater abundances on small versus large islands (Erwin *et al.* 1995). This may be attributed to the relationship between resource distribution in the patch and surrounding matrix (Estades 2001). Animals may retreat from the poor quality matrix into habitat patches (concentration effect), and then be reluctant to travel into the surrounding matrix (a so-called "fence effect"). However, Connor *et al.* (2007) found that animal population densities tend to be positively correlated with area, suggesting that density compensation may not be a common phenomenon.

Smaller patches, including plantings, have been found to contain bird communities with lower overall species richness and a greater proportion of generalist or edge-specialist species (Flaspohler *et al.* 2010; Mac Nally *et al.* 2010). Species that are tolerant to fragmentation may take advantage of nesting habitat provided by small patches while utilising resources in the surrounding matrix (Andrén 1994; Estades 2001; Driscoll *et al.* 2013). The superb fairywren accounted for over one quarter of all observations of breeding activity in our study, and is often described as a habitat generalist (Loyn *et al.* 2007; Mac Nally *et al.* 2010). Other commonly detected species, including the willie wagtail, demonstrated a positive relationship between breeding activity and patch size, indicating that not all species in the woodland assemblage respond similarly to patch size. Furthermore, we found no effect of patch size on the collective group of species of conservation concern.

Nest predation may have a significant influence on breeding success in birds, and can vary with predator type, patch size, and isolation in fragmented landscapes (Stephens *et al.* 2004;

Okada *et al.* 2017). There is conflicting evidence pertaining to the influence of patch size on nest predation in fragmented agricultural landscapes. For example, Hoover *et al.* (1995) attributed lower nesting success of wood thrushes (*Hylocichla mustelina*) in smaller fragments to a greater abundance of avian predators, and Major *et al.* (2001) found that the grey butcherbird (*Cracticus torquatus*), a predatory species in Australian woodlands, was more abundant in smaller than in larger habitat patches. In contrast, Zanette *et al.* (2000) found no evidence that area-sensitivity in the eastern yellow robin (*Eopsaltria australis*) could be explained by nest predation. Lehnen and Rodewald (2009) also found no evidence of area-sensitivity in survival and recruitment of shrubland bird species of conservation concern in the eastern United States. Nest type is also confounded with predation risk. Cup-nests are inherently more vulnerable to predation than dome-nests (Okada *et al.* 2017), and thus species that build cup-nests may be more sensitive to edge-effects in smaller patches. However, in our study, we found no evidence of a patch-size effect on cup-nesters. Conversely, smaller patches may contain lower abundances of brood parasites such as Horsfield's bronzecuckoo (*Chrysococcyx basalis*) (Brooker and Brooker 2003), reducing the risk of brood parasitism. Indeed, cuckoos were detected infrequently in our study sites (Appendix 2.2). Further research is warranted to directly investigate nesting success of woodland birds in fragmented agricultural landscapes.

A potential explanation for recording greater incidences of breeding activity in smaller patches than in larger patches is that an observer may search smaller sites more thoroughly than larger ones (Woolhouse 1983). However, we used a search method standardised by area and time in an attempt to control for potential effects of survey effort on activity detection rates. With an equivalent time spent per unit area in each survey regardless of patch size, bias towards detecting more breeding activity in smaller sites should not have influenced our

results. However, we note that breeding activity surveys are inherently biased towards species that nest in lower strata (such as the superb fairywren).

Patch type

We predicted that remnant woodland patches would be characterised by more cases of successful breeding than restoration plantings. However, our results showed that there was as much breeding activity in restoration plantings as in remnant woodland patches. This result is somewhat unexpected, as previous studies have found significant differences in bird species diversity and abundance in plantings and remnants; remnants, and large remnants in particular, tend to support a more diverse species assemblage than plantings (Arnold 2003; Loyn *et al.* 2007; Cunningham *et al.* 2008; Martin *et al.* 2011; Munro *et al.* 2011; Lindenmayer *et al.* 2012). Previous studies of bird assemblages in fragmented agricultural landscapes have identified bird species that are "planting specialists", which preferentially occupy restoration plantings over remnant woodland patches or other sites (reviewed by Belder *et al.* 2018). It was possible from the outset that breeding activity in restoration plantings would be primarily accounted for by a select few of these species, such as the generalist and edge-tolerant superb fairywren and willie wagtail. However, our modelling indicated that the same trend may hold for species of conservation concern as well as the woodland assemblage as a whole. This suggests that restoration plantings are providing habitat that is as valuable for bird populations as remnant woodland patches. We note, however, that various woodland-dependent species, including species of conservation concern such as the dusky woodswallow and brown treecreeper, show a strong affinity for remnant woodland. We therefore posit that restoration plantings play a complementary role in providing habitat for woodland birds, and caution against restoration plantings being considered a direct replacement for remnant woodland (see also Cunningham *et al.* 2007).

Patch shape

At this outset of this study, we predicted linear-shaped sites would support less breeding activity than block-shaped sites. We found a weak negative association between patch linearity and bird breeding activity in our study sites in only one candidate model, and therefore no strong evidence that site shape influenced bird breeding activity in our study region. Previous studies have suggested that increasing linearity negatively influences breeding birds (Helzer and Jelinski 1999; King *et al.* 2009). However, Selwood *et al.* (2009) found more evidence of successful breeding by woodland birds in linear patches. Our ordination modelling revealed that some bird species (e.g. brown treecreeper, black-faced cuckooshrike) showed more breeding activity in sites of increasing linearity, even though this was not reflected in relative abundance. We suggest that further studies are needed to confirm whether patch linearity influences breeding success of birds in fragmented agricultural landscapes.

Planting age

Contrary to expectations, we found that planting age was a negative predictor of bird breeding activity for the woodland assemblage and for species of conservation concern. That is, there was less evidence of breeding activity in older plantings. This finding contrasts with that of Selwood *et al.* (2009), who found that the age of plantings did not influence breeding activity. Barrett *et al.* (2003) found evidence of bird breeding activity in plantings as young as three years, noting that the species that exhibited the most breeding activity were small, shrub-swelling species such as the superb fairywren, red-browed finch, and yellow-rumped thornbill.

A typical planting in our study region consists of a *Eucalyptus* overstorey and *Acacia* understorey. In the absence of fire, an *Acacia* understorey is likely to senesce after 20-50 years (Broadhurst *et al.* 2008; Parsons and Gosper 2011), and natural regeneration of the shrub layer in planted sites may be poor (Vesk *et al.* 2008). The deterioration of understorey density and diversity with planting age is likely to contribute to a reduction in suitable nesting sites for common shrub-nesting species like the superb fairywren, as well as species of conservation concern such as the yellow-rumped thornbill and diamond firetail (*Stagonopleura guttata*). This may explain why the older plantings in our study, which were around 25 years of age, did not support as much breeding activity as younger plantings. The lack of an effect of age on the assemblage when our three most dominant species were removed, as well as the absence of effects for cup-nesters, may be related to the small sample size of these subsets.

Other findings

We found that examining breeding activity in our study sites provided a markedly different picture of bird species' responses to patch attributes than examining relative abundances obtained via point counts. There were several species whose responses to patch size, shape, and type based on relative abundance were opposite to their responses to these variables based on breeding activity. This indicates that 1) some bird species choose to breed disproportionately more in particular kinds of patches, or 2) the resources birds need to breed are not necessarily provided in patches that they choose to forage in (Loyn *et al.* 2007). The latter is of particular interest, and important for assessing the value of restoration plantings for woodland bird conservation; if birds preferentially occupy habitat patches but are unable to breed successfully in them, then those patches may become ecological traps, exacerbating population declines (Battin 2004). This highlights the importance of conducting research that

moves beyond pattern-based data collection to include more detailed, population-oriented studies (Ruiz-Jaen and Aide 2005; Belder *et al.* 2018).

We found that for species of conservation concern, there were no interpretable effects of site type, size, shape, or other variables on breeding activity score. The lack of an effect of site size is surprising, as previous studies have found that site occupancy by species of conservation concern is positively associated with patch size (Ford *et al.* 2009; Montague-Drake *et al.* 2009; Lindenmayer *et al.* 2010).

The absence of any effect of site type was also unexpected, as we had predicted more breeding activity by species of conservation concern in remnants due to the considerable body of evidence indicating that many threatened and declining species are dependent on or closely associated with remnant woodland (Kinross 2004; Cunningham *et al.* 2008; Martin *et al.* 2011). We note that some species of conservation concern, such as the yellow-rumped thornbill, are among "planting specialists" identified in previous studies (Belder *et al.* 2018) (Appendix 2.2). It is possible that the small number of observations of species of conservation concern in our study reduced our power to detect effects of patch attributes on these species, if they do indeed exist.

Inferential limitations

Variables at the landscape level, such as the amount and proximity of native vegetation, may have a stronger influence on species richness and abundance (Radford and Bennett 2007; Cunningham *et al.* 2008; Lindenmayer *et al.* 2010; Fahrig 2013) and breeding activity (Hinsley *et al.* 1995, 2008) than the patch-level characteristics of area and shape. Investigating these variables was outside the scope of this study, but we recommend further

research be undertaken to address their effects. We note the prevalence of a select few species in our data, which may be symptomatic of an environment that favours generalist and edge-tolerant species, to the detriment of richness and productivity of woodland bird assemblages in our study region. Additionally, the absence of the noisy miner in our study sites enabled us to examine the effects of patch attributes without the confounding effects of competitive exclusion, but noisy miners are regular occupants of small patches in fragmented agricultural landscapes (Major *et al.* 2001). We also note the small size (<10 ha) of plantings and remnants in our study. These reflect the typical size of native vegetation patches in our study region, but we caution against applying our findings to much larger-scale restoration projects, as breeding birds may respond differently to them than they do to small, isolated patches. Lastly, we note the relatively short duration of our study – two breeding seasons in years of above-average rainfall. We suggest that a better understanding of woodland bird population processes could be obtained by incorporating breeding studies into long-term monitoring projects.

Management implications and concluding remarks

Studies of bird distribution and abundance in fragmented landscapes have previously highlighted the conservation value of small habitat patches (Fischer and Lindenmayer 2002; Gibbons and Boak 2002; Flaspohler *et al.* 2010; Wintle *et al.* 2019). Our results add credence to these findings by providing evidence that birds not only occupy small patches, but display evidence of successful breeding within them. Previous studies of bird species richness and abundance in restoration plantings have recommended that plantings be as large as possible to maximise their conservation value (Freudenberger 2001; Watson *et al.* 2001; Westphal *et al.* 2007). We do not seek to undermine the conservation value of very large-scale restoration projects, which were outside the scope of this study, and we fully support the planting of large

areas of native vegetation as a strategy to increase vegetation cover in fragmented agricultural landscapes. However, our results suggest that the establishment and conservation of small plantings (and the conservation of small remnants) can also be of considerable value for the management of woodland bird populations (see also Schippers *et al.* 2009). It is often easier and more cost-effective to implement and maintain small patches (Kendal *et al.* 2017), so we are hopeful that our findings will encourage land managers to consider implementing small plantings wherever it is not possible to establish large plantings.

The observation of similar levels of breeding activity among the different site types in our study can be cautiously interpreted as encouraging for the conservation value of restoration plantings, as it indicates that birds in fragmented agricultural landscapes may view restoration plantings and remnant woodland patches as equally suitable breeding habitat. However, we acknowledge that breeding activity is only a proxy for breeding success, and cannot provide a true indication of whether breeding attempts are succeeding or failing. We therefore recommend further exploration using an approach such as monitoring nest success or daily nest survival.

Our finding that breeding activity decreased with planting age is of considerable interest for the management of restoration plantings. If a reduction in the condition and density of the shrub layer decreases the ability of a planting to support breeding birds, including species of conservation concern, then there is a case for active management of the shrub layer (including replanting if necessary) as a planting matures. Maintaining a complex habitat structure in restoration plantings also decreases the likelihood of colonisation by the noisy miner (Kinross and Nicol 2004; Maron *et al.* 2013). Although we did not find evidence that fenced sites supported more breeding activity, previous studies have shown that the ecological benefits of

restoration plantings are diminished when they are exposed to grazing by stock (Selwood *et al.* 2009; Lindenmayer *et al.* 2018b). We suggest that maintaining fences around restoration plantings may assist with preserving the shrub layer and ensuring that plantings continue to support breeding birds as they mature.

Finally, the unexpected nature of several of our key findings exemplifies the value of moving beyond pattern data (such as site occupancy information) towards a more behaviour- and population-oriented approach in monitoring and assessing the conservation value of restoration plantings and other habitat patches. As we have shown, relying solely on measures like species richness and abundance risks perpetuating critical knowledge gaps regarding habitat-use and the value of habitat patches for birds in fragmented agricultural landscapes.

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Appendices

Appendix 2.1 Attributes of study sites in the South-west Slopes bioregion.

site	site type	shape	planting year	elevation (m)	size (ha)	perimeter (m)	width (m)	fenced
WEBB-6	planting	block	2001	330	1.5	678	70	
SCHU-5	planting	block	2003	330	1.8	713	65	✓
BELL-4	planting	block	1990	270	2.5	1583	70	✓
HERI-3	planting	block	1990	285	5.3	983	150	✓
MART-1	planting	block	1991	280	5.6	988	200	✓
RETI-5	planting	block	2002	249	7.7	1176	200	✓
PALM-4	planting	linear	1997	269	1.3	1299	40	✓
PASS-3	planting	linear	1989	429	1.4	1054	40	✓
MATH-6	planting	linear	2000	344	1.7	1138	30	✓
FORR-A	planting	linear	1997	259	3.0	2610	15	✓
FORR-3	planting	linear	1997	300	3.2	1655	40	✓
MART-3	planting	linear	1993	276	3.2	2213	30	✓
STRO-1	remnant	block		303	2.1	1156	80	
WILS-3	remnant	block		325	2.8	1006	130	
PARK-2	remnant	block		265	5.8	1755	200	✓
WEBB-2	remnant	linear		262	2.3	1448	30	
SCHU-1	remnant	linear		297	4.1	736	60	
PARK-1	remnant	linear		248	3.6	2379	25	✓
GEDD-4	reference	block		397	47.1	3956	555	✓
KYEAMBA	reference	block		347	110	5070	400	✓
MATESG	reference	block		298	86	5090	400	✓

Appendix 2.2 Assemblages and attributes of bird species recorded during the study. Breeding activity scores are provided for each species and site type. The number of patches in which the species was detected and in which breeding occurred are provided in brackets: (no. patches breeding/no. patches present). • denotes species recorded in point count surveys but not breeding activity surveys. Species are listed in taxonomic order (Gill and Donsker 2018). Conservation status according to NSW threatened species listing (NSW Office of Environment and Heritage 2018) and bird atlas trends (Barrett et al. 2003). Categories are least concern (LC), conservation concern (CC), vulnerable (V). Information on breeding season and nest type taken from Morcombe (2003) and Pizzey and Knight (1997).

Species		Abbreviation	Nest type	Breeding season	Conservation status	Plantings	Remnants	Reference sites
stubble quail	*Coturnix pectoralis*	SQ	cup	Aug-Mar	LC	• (0/2)		• (0/1)
wedge-tailed eagle	*Aquila audax*	WTE	cup	Jun-Nov	LC			• (0/1)
brown goshawk	*Accipiter fasciatus*	BGOS	cup	Sep-Dec	LC	3.0 (1/0)	1.0 (1/0)	
nankeen kestrel	*Falco cenchroides*	NK	hollow	Aug-Dec	LC	1.0 (1/0)	• (0/1)	
brown falcon	*Falco berigora*	BRFA	cup	Aug-Nov	LC	• (0/2)	7.0 (1/1)	
painted buttonquail	*Turnix varius*	PBQ	cup	Aug-Feb	LC			• (0/1)
common bronzewing	*Phaps chalcoptera*	CBZ	cup	Aug-Dec	LC	6.0 (1/3)	6.0 (1/2)	
crested pigeon	*Ocyphaps lophotes*	CP	cup	Jul-Dec	LC	48.5 (3/3)	6.0 (2/5)	
peaceful dove	*Geopelia placida*	PD	cup	Oct-Jan	LC	2.0 (1/4)	12.0 (2/4)	
gang-gang cockatoo	*Callocephalon fimbriatum*	GGC	hollow	Oct-Jan	V			7.5 (1/1)
galah	*Eolophus roseicapilla*	GAL	hollow	Jul-Dec	LC	• (0/11)	• (0/6)	• (0/3)
little corella	*Cacatua sanguinea*	LCOR	hollow	Aug-Nov	LC	• (0/4)	• (0/3)	• (0/1)
sulphur-crested cockatoo	*Cacatua galerita*	SCC	hollow	Aug-Jan	LC	• (0/6)	• (0/4)	• (0/3)
crimson rosella	*Platycercus elegans*	CRO	hollow	Sep-Jan	LC	8.5 (2/7)		• (0/2)
eastern rosella	*Platycercus eximius*	ERO	hollow	Aug-Dec	LC	1.0 (1/12)	12.5 (2/6)	• (0/2)
red-rumped parrot	*Pseophotus haematonotus*	RRP	hollow	Aug-Jan	LC	4.0 (2/8)	• (0/5)	• (0/1)
Australian king-parrot	*Alisterus scapularis*	AKP	hollow	Sep-Jan	LC	• (0/1)	1.0 (1/1)	• (0/1)
superb parrot	*Polytelis swainsonii*	SUPA	hollow	Sep-Dec	V	• (0/2)	• (0/2)	
Horsfield's bronzecuckoo	*Chrysococcyx basalis*	HBC	parasitic	Aug-Jan	LC		• (0/1)	• (0/2)
shining bronzecuckoo	*Chrysococcyx lucidus*	SBC	parasitic	Aug-Jan	LC	• (0/1)		• (0/1)
pallid cuckoo	*Cacomantis pallidus*	PAC	parasitic	Aug-Dec	LC		• (0/1)	
fan-tailed cuckoo	*Cacomantis flabelliformis*	FTC	parasitic	Jul-Jan	LC	• (0/1)	• (0/1)	• (0/1)
laughing kookaburra	*Dacelo novaeguineae*	LK	hollow	Sep-Dec	LC	• (0/11)	11.0 (2/4)	5.0 (1/1)
sacred kingfisher	*Todiramphus sanctus*	SK	hollow	Sep-Jan	LC	• (0/2)	2.0 (2/3)	8.0 (1/2)
rainbow bee-eater	*Merops ornatus*	RBE	hollow	Oct-Jan	CC	• (0/3)	• (0/2)	• (0/1)
white-throated treecreeper	*Cormobates leucophaea*	WTTC	hollow	Aug-Jan	LC	• (0/1)	• (0/1)	1.0 (1/3)

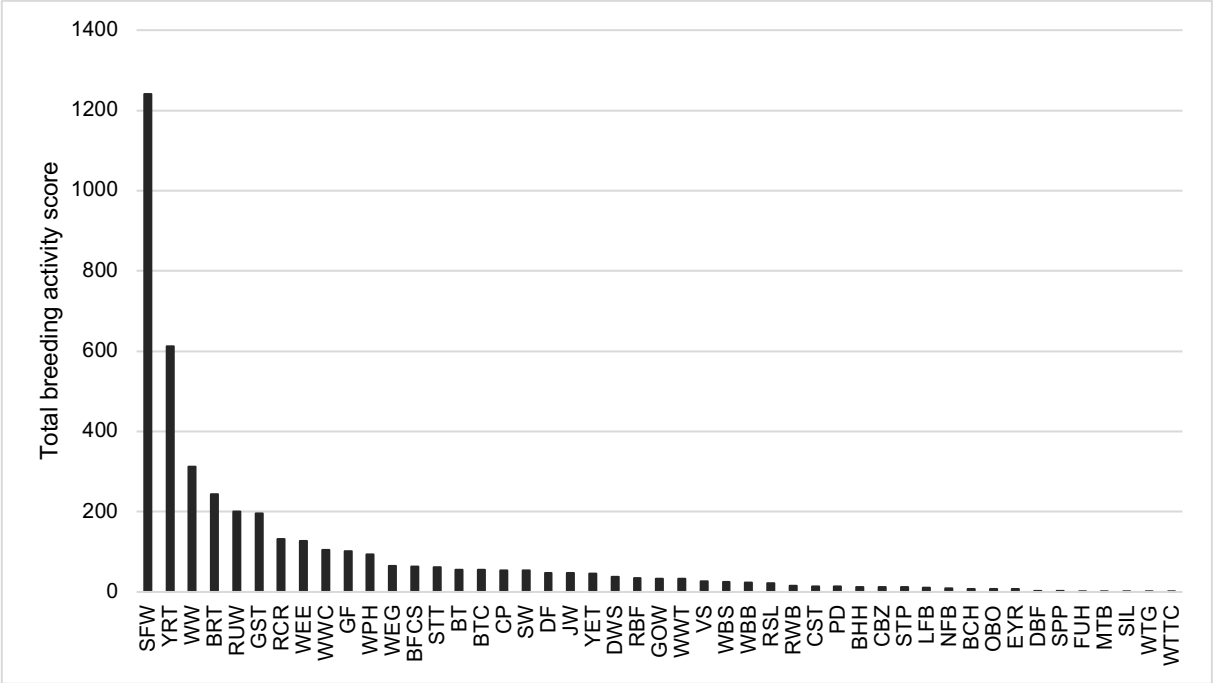
Species		Abbreviation	Nest type	Breeding season	Conservation status	Plantings	Remnants	Reference sites
brown treecreeper	*Climacteris picumnus*	BTC	hollow	May-Dec	V		29.0 (2/4)	26.0 (1/3)
superb fairywren[P]	*Malurus cyaneus*	SFW	dome	Sep-Dec	LC	831.5 (12/12)	262.5 (6/12)	146.5 (3/12)
little friarbird	*Philemon citreogularis*	LFB	cup	Jul-Nov	LC	10.0 (1/2)		
noisy friarbird	*Philemon corniculatus*	NFB	cup	Jul-Jan	LC	• (0/6)	• (0/1)	9.0 (1/3)
blue-faced honeyeater	*Entomyzon cyanotis*	BFH	cup	Jul-Jan	LC	• (0/2)		
black-chinned honeyeater	*Melithreptus gularis*	BCH	cup	Jul-Dec	V		7.5 (1/1)	• (0/2)
brown-headed honeyeater	*Melithreptus brevirostris*	BHH	cup	Aug-Jan	LC	• (0/5)	7.5 (1/2)	5.0 (1/3)
red wattlebird[P]	*Anthochaera carunculata*	RWB	cup	Jul-Dec	LC	15.0 (2/9)	• (0/3)	• (0/3)
yellow-faced honeyeater	*Caligavis chrysops*	YFH	cup	Jul-Jan	LC	• (0/3)		• (0/1)
noisy miner	*Manorina melanocephala*	NM	cup	Jul-Dec	LC	4.0 (2/6)	5.0 (2/3)	
fuscous honeyeater	*Ptilotula fusca*	FUH	cup	Aug-Dec	LC			1.0 (0/1)
white-plumed honeyeater[P]	*Ptilotula penicillata*	WPH	cup	Aug-Dec	LC	21.0 (3/11)	76.0 (3/6)	• (0/3)
spotted pardalote	*Pardalotus punctatus*	SPP	hollow	Sep-Dec	LC	2.0 (1/3)		• (0/1)
striated pardalote	*Pardalotus striatus*	STP	hollow	Jun-Jan	LC	11.0 (4/12)	1.0 (1/6)	• (0/3)
speckled warbler[P]	*Pyrrholaemus sagittatus*	SW	dome	Aug-Jan	CC	1.0 (1/1)	2.0 (1/2)	51.5 (3/2)
white-browed scrubwren[P]	*Sericornis frontalis*	WBS	dome	Jul-Dec	LC	25.0 (1/1)		
weebill[P]	*Smicrornis brevirostris*	WEE	dome	Aug-Feb	CC	97.0 (7/9)	7.0 (2/2)	23.0 (2/2)
western gerygone[P]	*Gerygone fusca*	WEG	dome	Aug-Nov	LC	26.5 (3/10)	2.0 (1/4)	36.5 (3/3)
white-throated gerygone	*Gerygone olivacea*	WTG	dome	Sep-Nov	LC	• (0/3)		1.0 (1/2)
brown thornbill	*Acanthiza pusilla*	BT	dome	Aug-Dec	LC	2.0 (2/1)	• (0/1)	54.0 (1/3)
buff-rumped thornbill	*Acanthiza reguloides*	BRT	dome	Aug-Dec	LC	32.5 (2/4)	55.5 (2/2)	155.5 (3/3)
yellow-rumped thornbill[P]	*Acanthiza chrysorrhoa*	YRT	dome	Jul-Dec	CC	513.5 (10/10)	53.0 (2/3)	48.5 (2/3)
yellow thornbill[P]	*Acanthiza nana*	YET	dome	Aug-Dec	LC	31.0 (8/9)	8.0 (2/2)	6.0 (2/3)
striated thornbill[P]	*Acanthiza lineata*	STT	dome	Jul-Dec	LC	17.5 (1/2)	13.0 (1/1)	31.5 (2/3)
white-browed babbler	*Pomatostomus superciliosus*	WBB	dome	Jun-Dec	LC	7.5 (1/2)	16.0 (1/1)	
grey butcherbird	*Cracticus torquatus*	GBB	cup	Aug-Dec	LC	• (0/2)	• (0/2)	
pied butcherbird	*Cracticus nigrogularis*	PBB	cup	Aug-Nov	LC	3.5 (1/10)	7.5 (1/5)	• (0/1)
Australian magpie	*Cracticus tibicen*	AM	cup	Aug-Oct	LC	143.5 (8/12)	74.5 (4/6)	19.5 (2/3)
pied currawong	*Strepera graculina*	PCW	cup	Aug-Dec	LC	• (0/2)	• (0/2)	• (0/1)
dusky woodswallow	*Artamus cyanopterus*	DWS	cup	Aug-Dec	V	1.0 (1/0)	36.0 (2/3)	6.0 (1/1)
black-faced cuckooshrike	*Coracina novaehollandiae*	BFCS	cup	Aug-Jan	LC	28.5 (4/12)	32.0 (4/5)	2.0 (2/3)
white-bellied cuckooshrike	*Coracina papuensis*	WBCS	cup	Aug-Mar	LC			• (0/1)
white-winged triller	*Lalage tricolor*	WWT	cup	Sep-Dec	CC	25.0 (2/1)	7.0 (1/2)	1.0 (1/2)

Species		Abbreviation	Nest type	Breeding season	Conservation status	Plantings	Remnants	Reference sites
varied sittella	*Daphoenositta chrysoptera*	VS	cup	Sep-Dec	V			27.0 (3/3)
crested shriketit	*Falcunculus frontatus*	CST	cup	Sep-Jan	CC	5.0 (3/5)	9.0 (1/4)	
golden whistler	*Pachycephala pectoralis*	GOW	cup	Aug-Jan	LC	23.5 (3/2)	2.0 (2/2)	7.5 (1/0)
rufous whistler[P]	*Pachycephala rufiventris*	RUW	cup	Sep-Feb	LC	164.5 (7/12)	16.5 (3/5)	20.0 (3/3)
grey shrikethrush[P]	*Colluricincla harmonica*	GST	cup	Jul-Feb	LC	135.0 (9/12)	52.5 (4/6)	8.5 (2/3)
olive-backed oriole	*Oriolus sagittatus*	OBO	cup	Sep-Jan	LC	7.5 (1/0)	• (0/1)	
willie wagtail[P]	*Rhipidura leucophrys*	WW	cup	Aug-Dec	LC	92.5 (9/12)	176.5 (5/6)	42.5 (1/3)
grey fantail[P]	*Rhipidura albiscapa*	GF	cup	Aug-Dec	LC	60.5 (9/12)	24.0 (3/5)	17.0 (3/3)
magpie-lark	*Grallina cyanoleuca*	AML	cup	Aug-Feb	LC	10.5 (4/12)	43.0 (2/6)	5.0 (1/3)
leaden flycatcher	*Myiagra rubecula*	LFC	cup	Sep-Nov	LC	• (0/1)		• (0/1)
restless flycatcher	*Myiagra inquieta*	RFC	cup	Aug-Jan	CC	• (0/2)	• (0/5)	• (0/1)
little raven	*Corvus mellori*	LR	cup	Aug-Dec	LC	16.5 (2/8)	7.5 (1/4)	9.0 (1/2)
Australian raven	*Corvus coronoides*	AR	cup	Jul-Oct	LC	11.5 (1/12)	3.5 (1/6)	11.5 (2/3)
white-winged chough	*Corcorax melanoramphos*	WWC	cup	Aug-Dec	LC	55.5 (6/6)	40.0 (2/3)	15.5 (2/3)
eastern yellow robin[P]	*Eopsaltria australis*	EYR	cup	Jul-Dec	LC		• (0/1)	7.0 (1/1)
hooded robin[P]	*Melanodryas cucullata*	HR	cup	Jul-Dec	V			4.5 (1)
jacky winter	*Microeca fascinans*	JW	cup	Jul-Dec	CC	• (0/1)	10.0 (2/2)	37.0 (2/2)
flame robin[P]	*Petroica phoenicea*	FR	cup	Aug-Jan	V	• (0/1)	• (0/1)	
red-capped robin[P]	*Petroica goodenovii*	RCR	cup	Jul-Jan	CC	95.5 (2/2)	• (0/1)	36.0 (3/1)
welcome swallow	*Hirundo neoxena*	WS	cup	Aug-Dec	LC	• (0/4)	• (0/6)	• (0/1)
fairy martin	*Petrochelidon ariel*	FM	other	Aug-Jan	CC	• (0/1)		
tree martin	*Petrochelidon nigricans*	TM	hollow	Aug-Dec	LC		8.5 (1/2)	
rufous songlark	*Cincloramphus mathewsi*	RSL	cup	Sep-Dec	LC	6.0 (1/8)	16.0 (3/3)	• (0/1)
brown songlark	*Cincloramphus cruralis*	BSL	cup	Sep-Feb	CC	• (0/3)	• (0/1)	
silvereye	*Zosterops lateralis*	SIL	cup	Sep-Jan	LC	1.0 (1/5)	• (0/2)	• (0/1)
common starling[I]	*Sturnus vulgaris*	STA	hollow	Aug-Jan		24.0 (2/8)	10.0 (1/5)	6.0 (1/1)
common blackbird[I]	*Turdus merula*	BKB	cup	Sep-Dec		42.0 (2/3)	• (0/1)	
mistletoebird	*Dicaeum hirundinaceum*	MTB	dome	Oct-Mar	LC	• (0/1)	• (0/2)	1.0 (1/2)
diamond firetail[P]	*Stagonopleura guttata*	DF	dome	Aug-Jan	V	38.0 (2/2)	10.0 (3/2)	
red-browed finch[P]	*Neochmia temporalis*	RBF	dome	Sep-Dec	LC	29.0 (2/3)	6.0 (1/1)	
double-barred finch	*Taeniopygia bichenovii*	DBF	dome	Jul-Dec	LC	2.0 (2/2)	1.0 (1/1)	
Australian pipit	*Anthus australis*	PIP	cup	Aug-Dec	LC	• (0/3)	• (0/2)	

[P] Planting specialists (Belder et al. 2018)

[I] Introduced species

Appendix 2.3 Total breeding activity recorded for the subset of bird species included in multivariate latent model ordinations. Acronyms corresponding to particular bird species are given in Appendix 2.2.



Appendix 2.4 Mixed effects models for breeding score modelled against weather and temporal variables, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-144.46	299.52	0.00	0.29
DATE + TIME	6	-143.57	300.00	0.49	0.23
DATE + SUN	6	-144.29	301.44	1.92	0.11
Intercept only	4	-160.21	328.82	29.30	0.00
Excluding superb fairywren	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-144.32	299.25	0.00	0.43
Intercept only	4	-150.68	309.75	10.50	0.00
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-145.52	301.64	0.00	0.29
DATE + TIME	6	-145.15	303.17	1.53	0.14
DATE + WIND	6	-145.30	303.46	1.82	0.12
Intercept only	4	-148.41	305.23	3.59	0.05
Species of conservation concern	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + TIME	6	-157.23	327.32	0.00	0.31
DATE	5	-158.89	328.39	1.06	0.18
Intercept only	4	-161.53	331.45	4.13	0.04
Cup-nesters	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + TIME	6	-150.37	313.60	0.00	0.23
DATE + TIME + WIND	7	-149.54	314.23	0.63	0.17
DATE + TIME + SUN	7	-149.66	314.48	0.88	0.15
DATE	5	-151.94	314.49	0.89	0.15
DATE + TIME + SUN + WIND	8	-148.96	315.42	1.82	0.09
DATE + SUN	6	-151.34	315.53	1.93	0.09
Intercept only	4	-155.94	320.29	6.69	0.01

Appendix 2.5 Mixed effects models for total breeding score modelled against site type for all sites (restoration planting, remnant, and reference), ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-144.46	299.52	0.00	0.54
DATE + FENCED	6	-144.27	301.40	1.89	0.21
Intercept only	4	-160.21	328.82	29.30	0.00
Excluding superb fairywren	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-144.32	299.25	0.00	0.67
Intercept only	4	-150.68	309.75	10.50	0.00
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-145.52	301.64	0.00	0.47
Intercept only	4	-148.41	305.23	3.59	0.08
Species of conservation concern	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-158.89	328.39	0.00	0.34
DATE + FENCED	6	-158.10	329.05	0.67	0.24
DATE + TYPE	7	-157.23	329.62	1.24	0.18
Intercept only	4	-161.53	331.45	3.07	0.07
Cup-nesters	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-151.94	314.49	0.00	0.60
Intercept only	4	-155.94	320.29	5.79	0.03

Appendix 2.6 Mixed effects models for total breeding score recorded during breeding activity surveys in restoration plantings and remnants (excluding reference sites), ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + SIZE	6	-122.29	257.58	0.00	0.22
DATE + SIZE + TYPE	7	-121.68	258.73	1.15	0.12
Intercept only	4	-140.70	289.88	32.29	0.00
Excluding superb fairywren	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-123.81	258.33	0.00	0.19
DATE + SIZE	6	-123.18	259.37	1.04	0.11
DATE + SIZE + SHAPE + SIZE:SHAPE	8	-121.17	260.12	1.79	0.08
Intercept only	4	-130.79	270.05	11.72	0.00
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-126.14	263.00	0.00	0.20
DATE + SIZE	6	-125.82	264.65	1.65	0.09
DATE + FENCED	6	-125.82	264.65	1.65	0.09
Intercept only	4	-129.16	266.79	3.79	0.03
Species of conservation concern	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-140.41	291.53	0.00	0.13
DATE + TYPE	6	-139.40	291.81	0.28	0.11
DATE + FENCED	6	-139.90	292.81	1.28	0.07
DATE + SIZE	6	-139.99	292.99	1.46	0.06
DATE + SIZE + TYPE	7	-138.84	293.05	1.52	0.06
DATE + SIZE + TYPE + SIZE:TYPE	8	-137.85	293.49	1.96	0.05
Intercept only	4	-143.08	294.63	3.10	0.03
Cup-nesters	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-133.19	277.10	0.00	0.23
DATE + TYPE	6	-132.92	278.85	1.75	0.10
DATE + SHAPE	6	-132.96	278.93	1.83	0.09
Intercept only	4	-136.61	281.69	4.59	0.02

Appendix 2.7 Mixed effects models for total breeding score recorded during breeding activity surveys in restoration plantings, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for the woodland assemblage, species of conservation concern, cup-nesters, and subsets of the woodland assemblage that exclude dominant species. All models that differed from the top model (ΔAIC_c) by ≤2 are shown, as well as the intercept-only model.

Woodland assemblage	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + AGE + SIZE + FENCED	8	-76.62	172.06	0.00	0.27
DATE + AGE + SIZE	7	-78.16	172.47	0.40	0.22
DATE + AGE + SIZE + SHAPE	8	-77.60	174.01	1.95	0.10
Intercept only	4	-92.38	193.49	21.43	0.00
Excluding superb fairywren	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + AGE	6	-79.79	173.16	0.00	0.20
DATE	5	-81.45	174.00	0.84	0.13
DATE + AGE + FENCED	7	-78.95	174.06	0.90	0.13
Intercept only	4	-84.72	178.18	5.01	0.02
Excluding superb fairywren, yellow-rumped thornbill, willie wagtail	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE	5	-82.35	175.81	0.00	0.22
Intercept only	4	-84.12	176.96	1.15	0.12
Species of conservation concern	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
DATE + AGE + SIZE	7	-90.30	196.75	0.00	0.20
DATE + AGE	6	-92.08	197.75	1.00	0.12
Intercept only	4	-96.31	201.35	4.59	0.02
Cup-nesters	df	log(*L*)	AIC_c	ΔAIC_c	AIC_w
Intercept only	4	-91.13	190.98	0.00	0.17
AGE	5	-90.42	191.94	0.96	0.10
DATE	5	-90.48	192.06	1.08	0.10

Appendix 2.8 Coefficients from multivariate latent variable models used to plot the effects of site attributes on relative abundance and breeding activity of bird species. The 95% upper and lower confidence limits around the estimate are provided in brackets. Estimates for which the confidence interval does not overlap zero are shown in bold.

	ABUNDANCE				BREEDING ACTIVITY			
common name	remnant vs. planting	reference vs. planting	patch size	patch linearity	remnant vs. planting	reference vs. planting	patch size	patch linearity
black-faced cuckooshrike	0.17 (-0.36, 0.80)	1.11 (-0.30, 2.61)	-0.21 (-0.76, 0.34)	-0.04 (-0.34, 0.21)	0.09 (-0.02, 0.21)	0.00 (-0.39, 0.32)	0.01 (-0.13, 0.14)	**0.05 (0.00, 0.09)**
brown-headed honeyeater	-0.67 (-1.86, 0.46)	-0.87 (-3.62, 1.59)	0.41 (-0.46, 1.42)	**-1.61 (-3.13, -0.59)**	0.00 (-0.04, 0.04)	**0.19 (0.06, 0.29)**	**-0.06 (-0.10, -0.01)**	0.00 (-0.01, 0.01)
buff-rumped thornbill	-0.19 (-1.25, 0.98)	-0.66 (-2.64, 1.36)	**1.28 (0.58, 2.03)**	**-1.49 (-2.86, -0.22)**	0.14 (-0.08, 0.36)	**1.11 (0.36, 1.76)**	-0.11 (-0.37, 0.18)	**-0.08 (-0.18, 0.00)**
brown thornbill	0.08 (-2.13, 1.96)	**3.63 (0.82, 6.17)**	-0.23 (-1.14, 0.71)	0.10 (-1.04, 1.15)	0.00 (-0.13, 0.11)	**1.67 (1.26, 2.13)**	**-0.50 (-0.65, -0.34)**	-0.01 (-0.05, 0.05)
brown treecreeper	**4.57 (2.56, 6.97)**	0.60 (-2.67, 4.10)	**1.50 (0.65, 2.42)**	0.09 (-0.32, 0.51)	**0.10 (0.04, 0.15)**	-0.01 (-0.19, 0.18)	0.02 (-0.04, 0.09)	**0.03 (0.01, 0.05)**
crested pigeon	**1.58 (0.53, 2.72)**	-0.56 (-6.02, 5.19)	-0.94 (-4.46, 1.42)	0.20 (-0.26, 0.68)	0.00 (-0.07, 0.07)	-0.03 (-0.23, 0.20)	0.00 (-0.08, 0.07)	0.01 (-0.02, 0.03)
double-barred finch	-0.29 (-1.94, 1.45)	-2.15 (-7.56, 2.84)	-0.50 (-3.18, 1.57)	**-1.48 (-3.00, -0.35)**	0.00 (-0.01, 0.01)	-0.01 (-0.05, 0.04)	0.00 (-0.01, 0.02)	0.00 (-0.01, 0.00)
diamond firetail	-0.71 (-3.20, 0.78)	-0.89 (-6.45, 3.72)	-1.35 (-4.36, 1.22)	**-5.67 (-7.29, -2.35)**	-0.05 (-0.15, 0.04)	-0.07 (-0.32, 0.23)	-0.01 (-0.11, 0.09)	**-0.03 (-0.06, 0.00)**
dusky woodswallow	**4.79 (2.91, 6.69)**	-2.01 (-6.44, 2.55)	1.59 (-0.11, 3.24)	-0.18 (-0.93, 0.69)	**0.10 (0.04, 0.16)**	-0.05 (-0.23, 0.17)	0.02 (-0.07, 0.08)	-0.02 (-0.04, 0.01)
grey fantail	**-1.01 (-1.56, -0.48)**	-0.05 (-1.29, 1.03)	0.14 (-0.27, 0.58)	**-0.46 (-0.76, -0.18)**	0.08 (-0.01, 0.18)	0.11 (-0.19, 0.41)	-0.02 (-0.13, 0.09)	**-0.05 (-0.09, -0.01)**
golden whistler	-0.47 (-2.93, 1.57)	-1.31 (-6.81, 3.88)	-1.98 (-4.89, 0.68)	-1.19 (-2.73, 0.41)	-0.03 (-0.15, 0.06)	-0.27 (-0.59, 0.04)	**0.12 (0.00, 0.13)**	-0.02 (-0.06, 0.02)
grey shrikethrush	-0.29 (-0.66, 0.13)	0.67 (-0.59, 1.96)	-0.44 (-0.94, 0.06)	-0.07 (-0.26, 0.13)	-0.04 (-0.21, 0.11)	-0.01 (-0.15, 0.19)	-0.04 (-0.23, 0.13)	0.00 (-0.06, 0.06)
jacky winter	**2.22 (0.93, 3.76)**	2.38 (-0.16, 4.64)	0.10 (-0.71, 0.89)	**-3.64 (-5.06, -2.38)**	0.06 (-0.04, 0.14)	-0.08 (-0.38, 0.21)	**0.11 (0.01, 0.23)**	-0.01 (-0.05, 0.02)
peaceful dove	**1.75 (0.92, 2.68)**	-0.68 (-5.18, 3.70)	-1.39 (-4.14, 0.59)	-0.21 (-0.71, 0.33)	**0.07 (0.01, 0.13)**	0.01 (-0.15, 0.19)	0.00 (-0.07, 0.07)	0.02 (-0.01, 0.04)
red-capped robin	**-2.71 (-4.69, -0.39)**	-3.03 (-7.72, 0.50)	0.59 (-1.33, 2.34)	**-3.71 (-5.39, -2.48)**	**-0.18 (-0.34, -0.02)**	-0.49 (-1.05, -0.04)	0.09 (-0.09, 0.30)	**-0.09 (-0.16, -0.03)**
rufous whistler	**-0.89 (-1.30, -0.48)**	0.63 (-0.38, 1.64)	-0.09 (-0.45, 0.29)	**-0.23 (-0.43, -0.02)**	**-0.16 (-0.33, -0.03)**	-0.13 (-0.56, 0.35)	0.01 (-0.17, 0.17)	0.03 (-0.02, 0.10)
red wattlebird	-0.49 (-1.06, 0.13)	**-3.48 (-6.60, -1.03)**	**1.12 (0.31, 2.01)**	0.01 (-0.24, 0.30)	-0.02 (-0.06, 0.02)	-0.01 (-0.15, 0.11)	0.00 (-0.05, 0.05)	0.01 (-0.01, 0.03)
sacred kingfisher	**1.78 (0.84, 2.85)**	-1.34 (-4.55, 1.52)	**1.17 (0.25, 2.17)**	**-0.90 (-1.82, -0.03)**	0.01 (-0.05, 0.06)	**-0.17 (-0.37, 0.00)**	**0.10 (0.03, 0.17)**	0.00 (-0.02, 0.03)
striated pardalote	0.11 (-0.33, 0.62)	-2.27 (-4.85, 0.14)	0.46 (-0.28, 1.48)	0.13 (-0.06, 0.35)	-0.01 (-0.04, 0.01)	-0.01 (-0.09, 0.07)	0.00 (-0.03, 0.03)	0.00 (-0.01, 0.01)
striated thornbill	-0.61 (-2.61, 1.19)	**3.52 (1.05, 6.37)**	-0.12 (-1.24, 0.81)	**-2.06 (-4.95, -0.29)**	0.03 (-0.12, 0.17)	-0.02 (-0.46, 0.37)	0.09 (-0.06, 0.24)	-0.02 (-0.07, 0.03)
speckled warbler	-1.18 (-3.06, 0.41)	1.19 (-1.81, 4.39)	-0.11 (-1.32, 1.13)	**-2.69 (-4.29, -1.07)**	0.00 (-0.13, 0.13)	0.06 (-0.30, 0.48)	0.14 (-0.01, 0.28)	0.00 (-0.05, 0.05)

	ABUNDANCE				BREEDING ACTIVITY			
common name	remnant vs. planting	reference vs. planting	patch size	patch linearity	remnant vs. planting	reference vs. planting	patch size	patch linearity
varied sittella	-1.65 (-4.50, 1.29)	3.10 (-0.10, 6.34)	0.14 (-0.84, 1.23)	-1.54 (-4.17, 0.57)	0.00 (-0.08, 0.08)	0.06 (-0.16, 0.30)	0.05 (-0.03, 0.13)	0.00 (-0.03, 0.03)
white-browed scrubwren	-3.00 (-7.08, 0.28)	-2.34 (-7.36, 3.75)	**-4.09 (-8.19, -0.80)**	-0.77 (-3.26, 1.14)	-0.05 (-0.14, 0.04)	-0.05 (-0.30, 0.27)	-0.01 (-0.11, 0.13)	-0.01 (-0.05, 0.02)
weebill	**-0.97 (-1.57, -0.33)**	**-5.07 (-6.81, -3.25)**	**1.90 (1.24, 2.47)**	**-0.67 (-1.01, -0.38)**	-0.13 (-0.27, 0.01)	-0.32 (-0.80, 0.12)	0.08 (-0.08, 0.26)	-0.04 (-0.09, 0.02)
western gerygone	**-0.91 (-1.62, -0.18)**	0.16 (-1.39, 1.56)	-0.01 (-0.60, 0.52)	**-0.88 (-1.30, -0.50)**	-0.06 (-0.17, 0.05)	-0.11 (-0.45, 0.25)	0.11 (-0.02, 0.24)	-0.02 (-0.06, 0.02)
white-plumed honeyeater	**0.77 (0.34, 1.12)**	**-4.75 (-7.35, -2.30)**	**1.51 (0.77, 2.36)**	0.09 (-0.10, 0.31)	**0.14 (0.03, 0.24)**	-0.01 (-0.34, 0.28)	0.00 (-0.11, 0.13)	0.02 (-0.02, 0.06)
willie wagtail	**0.53 (0.15, 0.91)**	-0.49 (-1.95, 1.13)	0.07 (-0.43, 0.63)	0.04 (-0.15, 0.24)	**0.47 (0.29, 0.66)**	-0.42 (-1.03, 0.17)	**0.22 (0.02, 0.45)**	0.03 (-0.05, 0.02)
white-winged chough	-0.52 (-1.40, 0.55)	-0.73 (-3.26, 1.75)	0.46 (-0.36, 1.36)	**-1.34 (-2.19, -0.62)**	0.03 (-0.07, 0.13)	-0.01 (-0.31, 0.28)	0.04 (-0.06, 0.16)	-0.01 (-0.05, 0.02)
yellow thornbill	**-1.39 (-2.05, -0.74)**	0.03 (-1.55, 1.66)	-0.30 (-0.98, 0.34)	**-0.39 (-0.68, -0.13)**	-0.04 (-0.11, 0.05)	-0.15 (-0.38, 0.09)	0.07 (-0.02, 0.15)	0.00 (-0.03, 0.03)
yellow-rumped thornbill	**-1.66 (-2.29, -1.08)**	0.07 (-1.53, 1.49)	-0.44 (-1.07, 0.19)	-0.19 (-0.42, 0.04)	**-0.54 (-0.84, -0.24)**	-0.59 (-1.59, 0.38)	0.00 (-0.39, 0.34)	-0.07 (-0.19, 0.04)

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CHAPTER 3

Revegetation and reproduction: Do restoration plantings in agricultural landscapes support breeding populations of woodland birds?

Hooded robin (*Melanodryas cucullata*) nest and egg. Photo: Donna Belder.

Belder, D. J., Pierson, J. C., Ikin, K., and Lindenmayer, D. B. (2020) Revegetation and reproduction: Do restoration plantings in agricultural landscapes support breeding populations of woodland birds? *Oecologia* **192**, 865–878: Highlighted Student Research.

Abstract

Restoration plantings are frequently occupied by native wildlife, but little is known about how planting attributes influence breeding by, and persistence of, fauna populations. We monitored breeding success of woodland birds in restoration plantings in a fragmented agricultural landscape in south-eastern Australia. We documented nest fate and daily nest survival (DSR) in plantings and remnant woodland sites. We analysed the influence on breeding success of patch attributes (size, shape, type) compared to other potentially influential predictors such as nest-site and microhabitat variables. We found that, in general, patch attributes did not play a significant role in determining breeding success for woodland birds. However, we examined a subset of species of conservation concern, and found higher DSR for these species in restoration plantings than in similarly sized woodland remnants. We also found negative effects of patch size and linearity on DSR in species of conservation concern. The primary cause of nest failure was predation (91%). We used camera trap imagery to identify the most common nest-predators in our study sites: native predatory bird species, and the introduced red fox (*Vulpes vulpes*). Our findings are further evidence of the value of restoration plantings and small habitat patches for bird populations in fragmented agricultural landscapes. We recommend controlling for foxes to maximise the likelihood that restoration plantings and other woodland patches in Australia support breeding populations of woodland birds. More broadly, our study highlights the importance of taking a detailed, population-oriented approach to understanding factors that influence habitat suitability for fauna of conservation concern.

Keywords: revegetation, temperate woodland, SLOSS, population dynamics

Introduction

Habitat loss due to agricultural expansion is a key threat to biodiversity in many parts of the world (Maxwell *et al.* 2016; Egli *et al.* 2018). Ongoing loss of habitat in fragmented agricultural landscapes is making it increasingly difficult for many organisms to persist and maintain viable populations (Bennett *et al.* 2015; Haddad *et al.* 2015; Stanton *et al.* 2018). In an attempt to address these problems, there are increasing efforts to replant native vegetation in agricultural landscapes in many parts of the world. Ecological tree plantings – hereafter referred to as "restoration plantings" – collectively comprise millions of hectares of planted vegetation, costing billions of dollars to establish and maintain (Kimball *et al.* 2015; Crouzeilles *et al.* 2016). They are often implemented as a specific conservation strategy to replace lost habitat for threatened and declining fauna (McAlpine *et al.* 2016; Catterall 2018; Lindenmayer *et al.* 2018*a*). For example, in south-eastern Australia, over 90% of box-gum grassy woodland habitat has been lost (Thiele and Prober 2000), and woodland birds in this region have suffered substantial population declines (Barrett *et al.* 2007; Rayner *et al.* 2014). Consequently, woodland birds are frequently considered among the key beneficiaries of restoration plantings in south-eastern Australia (Belder *et al.* 2018).

There is evidence suggesting that many species of woodland bird will readily occupy restoration plantings (Barrett *et al.* 2008; Lindenmayer *et al.* 2010; Debus *et al.* 2017). Studies examining woodland bird responses to restoration plantings typically use pattern data such as presence and abundance to infer habitat quality. Previous research has offered insights into colonisation and extinction patterns (Barrett *et al.* 2008; Mortelliti and Lindenmayer 2015), changes in bird community composition in plantings over time (Mac Nally 2008; Lindenmayer *et al.* 2016, 2018*c*; Debus *et al.* 2017), and occupancy trends relating to site type, habitat structure, and composition (Martin *et al.* 2011; Munro *et al.* 2011; Ikin *et al.*

2018). However, do patch attributes have the same influence on breeding success as they do on site occupancy? Few studies have investigated breeding success in restoration plantings, and little is known about the role of restoration plantings in supporting successful breeding by woodland birds.

The presence of a species in a restoration planting does not necessarily mean that the site is supporting successful breeding of that species. Previous work has found that the relative abundance of woodland bird species is not necessarily correlated with their degree of breeding activity (Belder *et al.* 2019). For restoration plantings to support breeding populations of woodland birds, they must provide adequate resources and quality habitat to encourage persistence of individuals in a site, and to enable resident individuals to breed successfully (Arlt and Pärt 2007; Flockhart *et al.* 2016). This is an important outcome if restoration plantings are to be widely implemented as a conservation strategy (Ruiz-Jaen and Aide 2005).

In this study, we focus on breeding success as an indicator of habitat quality in restoration plantings and remnant woodland patches. Breeding success is a key measure of the productivity and quality of a habitat patch (Hinsley *et al.* 2008; Milligan and Dickinson 2016). By assessing whether successful breeding is occurring, we can begin to assess the extent to which a habitat patch is supporting the species that it is intended to help conserve. Moreover, identifying site attributes (e.g. size, shape) that best support successful breeding facilitates conservation planning, and has the potential to improve the cost-effectiveness of restoration plantings as a conservation strategy.

It is also important to identify the most common causes of breeding failure in restoration plantings. For example, low nesting success could be due to an introduced predator that thrives in fragmented agricultural landscapes, such as the red fox (*Vulpes vulpes*) in Australia (Braysher 2017). If this is the case, then an otherwise good quality restoration planting may never support species that are vulnerable to fox predation. However, this type of threat, once identified, could be readily addressed in management plans. Conversely, if nest predation is low but birds are abandoning nests or failing to fledge their young, it may indicate that resource limitation is the primary factor influencing breeding success and survival (Zanette *et al.* 2000). In this case, more detailed studies might establish what is driving resource limitation. For example, a lack of suitable nesting sites (exposure, competition), food shortage, or perhaps inefficient foraging strategies due to home ranges that are constrained by patch geometry or landscape context.

Research objectives

The primary aim of this study was to determine whether restoration plantings are able to support breeding populations of woodland birds. We used two different indicators of breeding success: nest fate and daily nest survival. Specifically, we posed the following three questions:

Question 1. How does breeding success in restoration plantings compare to breeding success in remnant woodland patches?

We compared breeding success (nest fate and daily nest survival) in restoration plantings and similarly sized woodland remnants. We used larger woodland remnants, such as travelling stock reserves, as "reference" sites. Belder *et al.* (2019) found equal levels of breeding

activity in restoration plantings and woodland remnants in the study area. We therefore predicted that breeding success in plantings would be similar to that in remnants.

Question 2. Are patch attributes such as size, shape and type important determinants of breeding success in plantings and remnant woodland patches?

We used a model selection approach to compare the influence of patch attributes (size, shape, type) with other variables that may influence breeding success, including nest-site variables (distance to edge of patch, height off ground, concealment), and microhabitat variables (shrub cover, ground layer composition). A previous study by Belder *et al.* (2019) identified a negative relationship between patch size and breeding activity, and a positive relationship between planting age and breeding activity. We expected these findings to be reflected in our study of breeding success, and postulated that patch attributes would significantly influence breeding success.

Question 3. What are the primary causes of nest failure in restoration plantings and woodland remnants?

We sought to identify the reasons for nest failure in restoration plantings, and establish whether the same processes are responsible for nest failure in woodland remnants. We predicted that predation would be the leading cause of nest failure in all sites, as it is the primary driver of nest failure in most bird communities (Belder *et al.* 2018). We also sought to quantify whether major nest-predators differ between patch types. Based on research conducted in a similar study region (Okada *et al.* 2017), and a recent review of nest-predators in Australia (Fulton 2019), we expected the dominant predators of woodland bird nests to be predatory bird species, including ravens (*Corvus* spp), butcherbirds (*Cracticus* spp), and currawongs (*Strepera* spp). We also expected the eastern brown snake (*Pseudonaja textilis*) to

be a common nest-predator in restoration plantings, as they have been detected more frequently in plantings than in similarly sized woodland remnants in our study region (Cunningham *et al.* 2007). Snakes have been identified as important nest-predators in Australia (Fulton 2019) and internationally (Weatherhead and Blouin-Demers 2004).

Approach

Our study was conducted over two breeding seasons and used real, active bird nests to quantify breeding success and nest-predation. Previous studies in Australian landscapes, including in our study region, have used indicators of breeding activity as a proxy for breeding success (Barrett *et al.* 2008; Selwood *et al.* 2009; Mac Nally *et al.* 2010; Belder *et al.* 2019). While such indirect measures are an important step away from traditional diversity and abundance measures, they cannot accurately represent breeding success or identify reasons for breeding failure. We document, for the first time, nesting success, daily nest survival, and primary predators of woodland birds breeding in restoration plantings in a fragmented agricultural landscape.

Materials and Methods

Study area

We conducted our study in the South-west Slopes bioregion of New South Wales, Australia. The region is part of Australia's sheep-wheat belt and has been extensively cleared of native vegetation, with as little as 0.1% of the original temperate woodland remaining in intact condition (Thiele and Prober 2000). Remnant patches are predominantly white box (*Eucalyptus albens*) / yellow box (*E. melliodora*) / Blakely's red gum (*E. blakelyi*) grassy woodland, a critically endangered ecological community (NSW OEH 2016). Patches of red

stringybark (*E. macrorhyncha*) woodland and mugga ironbark (*E. sideroxylon*) woodland are also present.

Study sites

We used spring bird survey data collected over 12 years (see Lindenmayer *et al.* 2018*c*) to select a subset of 21 long-term monitoring sites: 12 plantings (1.3-7.7 ha), six similarly sized woodland remnants (2.1-5.8 ha), and three large, intact remnants ("reference" sites >44 ha) (Figure 3.1). Plantings were aged between 12 and 25 years. We attempted to control for the effects of competitive exclusion by selecting sites that did not have a history of occupancy by the noisy miner (*Manorina melanocephala*). Details regarding study site selection are described in Belder *et al.* (2019).

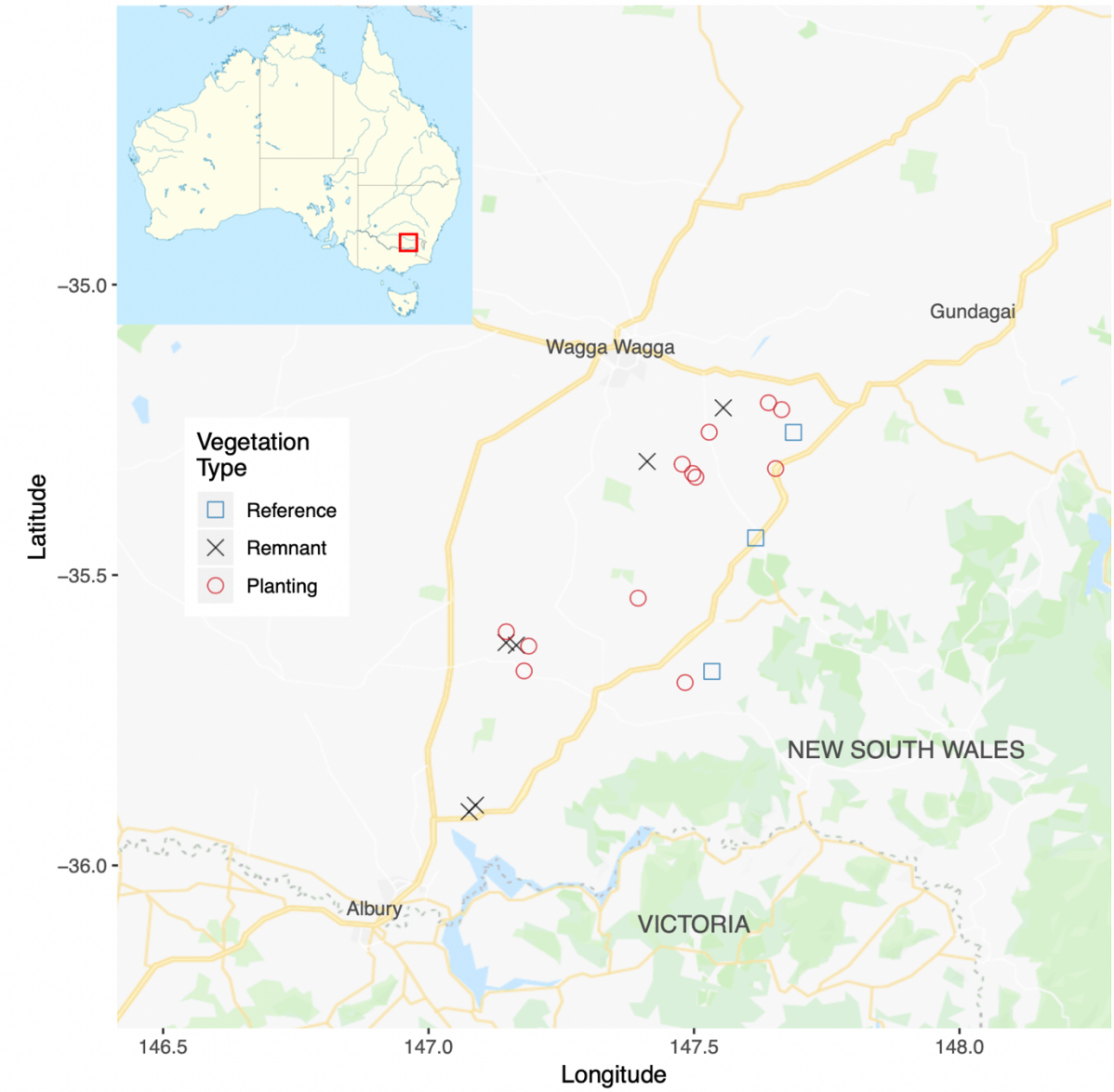

Figure 3.1 Map of study sites in the South-west Slopes bioregion of New South Wales, Australia. Map created using ggmap for R (Kahle and Wickham 2013).

Nest searches

We conducted fixed time-per-unit-area surveys (one hour per hectare) to locate nests in study sites over two breeding seasons. We completed two rounds of surveys (October and November) in 2015, and three rounds (September, October, November) in 2016. We searched sites systematically, with search areas designated by size and shape of sites. For sites with a total area less than 3 ha, we searched 1.3 ha within the site – this was equivalent to the size of the smallest site in the study. For sites with a total area greater than 3 ha, we searched 3 ha

within the site. We surveyed block sites in a grid fashion, and linear sites along their length until we had searched the desired area (i.e. 1.3 ha or 3 ha). Due to the large geographic spread of sites, we were unable to completely randomise the order of site surveys during each round. However, we ensured that sites were not consistently surveyed at the same time of day. Sites were surveyed at any time of day from dawn to dusk, except during November 2016, when sites were surveyed only in the four hours post-sunrise and pre-sunset.

Nest monitoring

Once a nest was located, we used flagging tape to mark its position (near to but not at the nest to avoid attracting the attention of predators) and recorded its location using a handheld GPS. Depending on accessibility, we determined the status (i.e. the stage of development) of the nest at discovery by either manual inspection or through observations of parental behaviour. Some nests required multiple visits on different days to ascertain status. We conducted regular checks in person to verify status – every 7-10 days in 2015, and every 3-5 days in 2016. We inspected nests manually or used a nest inspection tool (endoscopy-type camera for dome nests, and mirror on an extendible pole for open cup nests). For nests that were out of arm's reach or could not be reached by extendible pole, we used behavioural observations to determine status. We observed nests for up to 30 minutes, or until we recorded activity at the nest and could verify the status. If we could not determine the status within the 30 minute observation period, we repeated the observation at the next scheduled visit (3-5 days later in 2016). If we did not record activity in three consecutive visits, we assumed the nest was no longer active. In the later stages of nesting (i.e. when the nest was estimated to be within 5 days of fledging), we did not approach the nest, and used only behavioural observations to determine status. This was to minimise the risk of premature fledging. We considered a nesting attempt to have succeeded if at least one chick fledged.

Where possible, we used fixed motion-sensing wildlife cameras to monitor nests, with the primary aim of detecting nest predation. We used a combination of Bushnell Trophy HD, UOVision UV565HD, and HCO ScoutGuard SG560K black flash cameras. All cameras are triggered by motion within the field of view. To reduce the incidence of false triggers (e.g. by wind-blown foliage), we set camera sensitivity to "low". We were able to use nest cameras for nests at heights of up to 6 m.

Nest site measurements and microhabitat surveys

For all nests, we recorded a GPS location (accurate to the nearest 2 m), the height of the nest above ground, and the substrate (foliage, branch, woody debris, etc.) in which the nest was built. For nests in the 2016 breeding season, we also recorded concealment (visually estimated at a distance of approximately 10 m, and to the nearest 5%). We used ArcMap (ESRI 2018) to calculate the distance of each nest to the nearest patch edge. Where relevant, we also recorded the plant species in which nests were built.

For nests in the 2016 breeding season, we collected microhabitat data around the nest site. We conducted microhabitat surveys when the nest was no longer active (either fledged or failed). At each nest, we used a tape measure to mark out a quadrat measuring 25 m along its diagonal, with the nest at its centre. The two diagonals were aligned north-south and east-west. We visually estimated the proportion of ground cover (to the nearest 1%) and midstorey cover (to the nearest 5%). We chose these microhabitat variables as multiple studies have documented their influence on site occupancy by woodland birds (Seddon *et al.* 2003; Antos and Bennett 2006; Montague-Drake *et al.* 2009; Munro *et al.* 2011).

Statistical analyses

We used a model selection approach (Burnham and Anderson 2004) to investigate the effects of patch-level, nest-level, and microhabitat variables on nest fate and daily nest survival (Table 3.1). For clarity, and to address inconsistencies with data collection between years, we modelled data only from nests monitored in 2016. We used generalised linear mixed effects regression models with study site as a random effect. Our response variables were nest fate (binary, where success = 0 and fail = 1), and daily nest survival (DSR). For nest fate and DSR analyses, we excluded nests for which the failure date was uncertain (to the nearest five days), most of which were classified as "abandoned". We included these nests, along with those monitored in 2015, when calculating the total proportion of successful nests, and we report these results in the *General findings* section of our Results. Due to inherent differences in nest survival, we analysed dome-nesters and cup-nesters separately. We had sufficient data to individually examine one dome-nesting species: the superb fairywren (*Malurus cyaneus*), and one cup-nesting species: the willie wagtail (*Rhipidura leucophrys*). We also examined a subset of dome-nesting species of conservation concern (Appendix 3.1). We did not include nests of introduced species in our study.

We used a comparative model selection approach, in which we modelled combinations (sets) of variables and used Akaike's Information Criterion to determine which variables best predicted nest fate and DSR:

4. Patch attributes: type, size (ha), shape (calculated as perimeter/width)
5. Nest site attributes: height above ground, distance to edge of patch, concealment
6. Microhabitat variables: shrub cover and ground layer composition within 20 m of the nest.

Table 3.1 Linear mixed model parameters. The response variables are FATE and DSR, and all other variables are predictors.

Variable name	Response/predictor	Model set	Description
FATE	Response		Nest fate (a binary variable where 0 = survive and 1 = fail)
DSR	Response		Daily survival rate, calculated using Program MARK
DATE	Predictor		Julian date of nest discovery
TYPE	Predictor	Patch	Patch type (planting, remnant, reference)
SIZE	Predictor	Patch	Patch size (ha)
SHAPE	Predictor	Patch	Measure of patch shape, calculated as perimeter/width (m)
AGE	Predictor	Age	Age of planting at the commencement of the study (years)
HEIGHT	Predictor	Nest	Height of nest above ground (m)
DIST_EDGE	Predictor	Nest	Distance of nest to nearest patch edge (m)
CONCEALMENT	Predictor	Nest	Nest concealment, estimated at approx. 10 m from the nest (%)
BARE GROUND	Predictor	Microhabitat	Proportion of bare ground cover within 20 m of the nest
LEAF LITTER	Predictor	Microhabitat	Proportion of leaf litter cover within 20 m of the nest
GRASS	Predictor	Microhabitat	Proportion of exotic grass cover within 20 m of the nest
WOODY DEBRIS	Predictor	Microhabitat	Proportion of woody debris cover within 20 m of the nest
SHRUB COVER	Predictor	Microhabitat	Amount of midstorey shrub cover (%)

We included date of nest discovery (DATE) as an explanatory variable in all models, as preliminary analyses indicated that date within the breeding season was a significant influence on breeding success. For the first two sets of variables, we fitted models with the variables of interest plus interaction terms. For models including microhabitat variables, we did not include interaction terms. For daily nest survival model selection, we included a model that assumed constant nest survival (null model). We ranked candidate models using Akaike's Information Criterion corrected for small sample sizes (AIC_c). We considered models with $\Delta AIC_c \leq 2$ as top-ranked models (Burnham and Anderson 2004).

When reporting DSR results, we provide both the sample size (n) and effective sample size (n_{ess}) (Rotella *et al.* 2004; Shaffer and Thompson 2007). Effective sample size is equal to the number of known days survived for each nest plus the number of intervals in which a nest

failed (Rotella *et al.* 2004). For example, a nest that survived for 10 days and then failed between day 10 and day 13 contributes 11 to the study's effective sample size.

We used the packages 'lme4' (Bates *et al.* 2015) and 'MuMIn' (Bartoń 2018) in R version 3.5.2 (R Core Team 2019) to fit and select models for FATE. For DSR calculation and model selection, we used Program MARK (White and Burnham 1999) via the R package 'RMark' (Laake 2003).

Prior to fitting models, we checked all explanatory variables for multi-collinearity using variance inflation factors. We corrected for multi-collinearity by removing large reference sites from models that included both size and type (site type was significantly correlated with site size due to the comparatively large size of reference sites). That is, we included data only from plantings and similarly sized woodland remnants when modelling our response variable against site size and shape. We also scaled and centred our continuous predictor variables for generalised linear mixed modelling.

Results

General

We located 324 woodland bird nests over the course of the two years of field study: 89 in 2015, and 235 in 2016. Of these, we were able to successfully track the fate of 222 nests, or 69% of the total number of nests. Of the nests that were tracked successfully, 129 were in plantings (12 sites), 45 were in remnants (six sites), and 48 were in large reference sites (three sites). We analysed nests from 24 different woodland bird species: 11 dome-nesters and 13 cup-nesters (Appendix 3.1).

Nests were predominately in the lower strata. Mean nest height was 2.2 m (SE=0.16 m). Cup nests in large reference sites were significantly higher on average than in restoration plantings and similarly sized woodland remnants ($p<0.0001$). Site type did not influence nest height for dome nests.

Mean nest success (succeed vs. fail) across all nest types was 33.8%. Success rates were 29.6% for cup-nesters and 38.1% for dome-nesters (Figure 3.2).

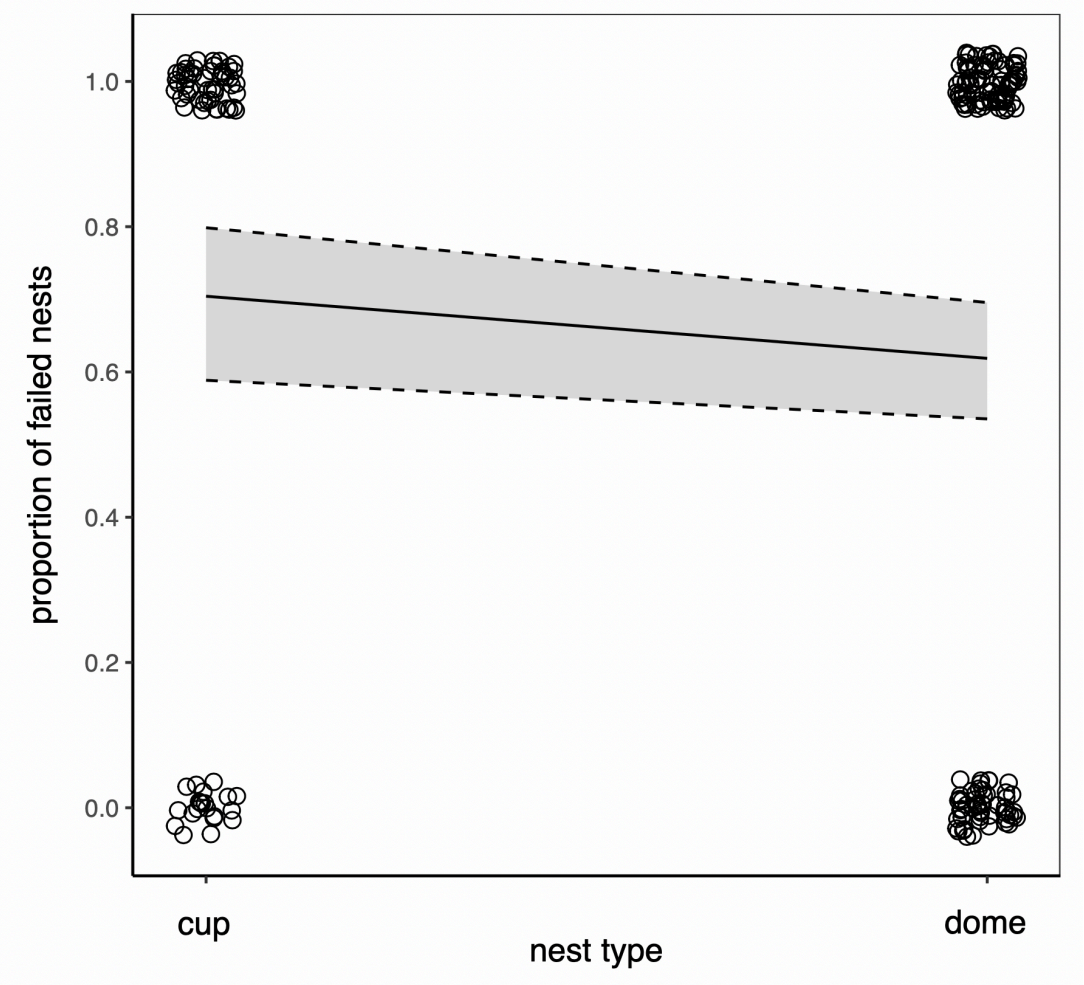

Figure 3.2 Proportion of failed woodland bird nests according to nest type. Shaded areas indicate upper and lower 95% confidence intervals. Clustered points indicate frequency of success (0) and failure (1) for each nest type. Data from both 2015 and 2016 were modelled to produce estimates. Plot created using ggplot2 for R (Wickham 2016).

We calculated daily nest survival for 107 dome nests (n_{ess} = 2134) and 50 cup nests (n_{ess} = 599) (Table 3.2). As the breeding season progressed, DSR decreased for dome-nesters but increased for cup-nesters (Figure 3.3).

Table 3.2 Number of nests (*n*) and effective sample size (n_{ess}) used to calculate daily nest survival (DSR) for each subset of the woodland bird assemblage.

Subset	Sites	*n*	n_{ess}
dome	planting + remnant + reference	107	2134
	planting + remnant	86	1682
	planting	72	1393
cup	planting + remnant + reference	50	599
	planting + remnant	39	428
superb fairywren	planting + remnant + reference	56	1046
	planting + remnant	46	826
	planting	37	652
conservation concern	planting + remnant + reference	34	720
	planting + remnant	31	647

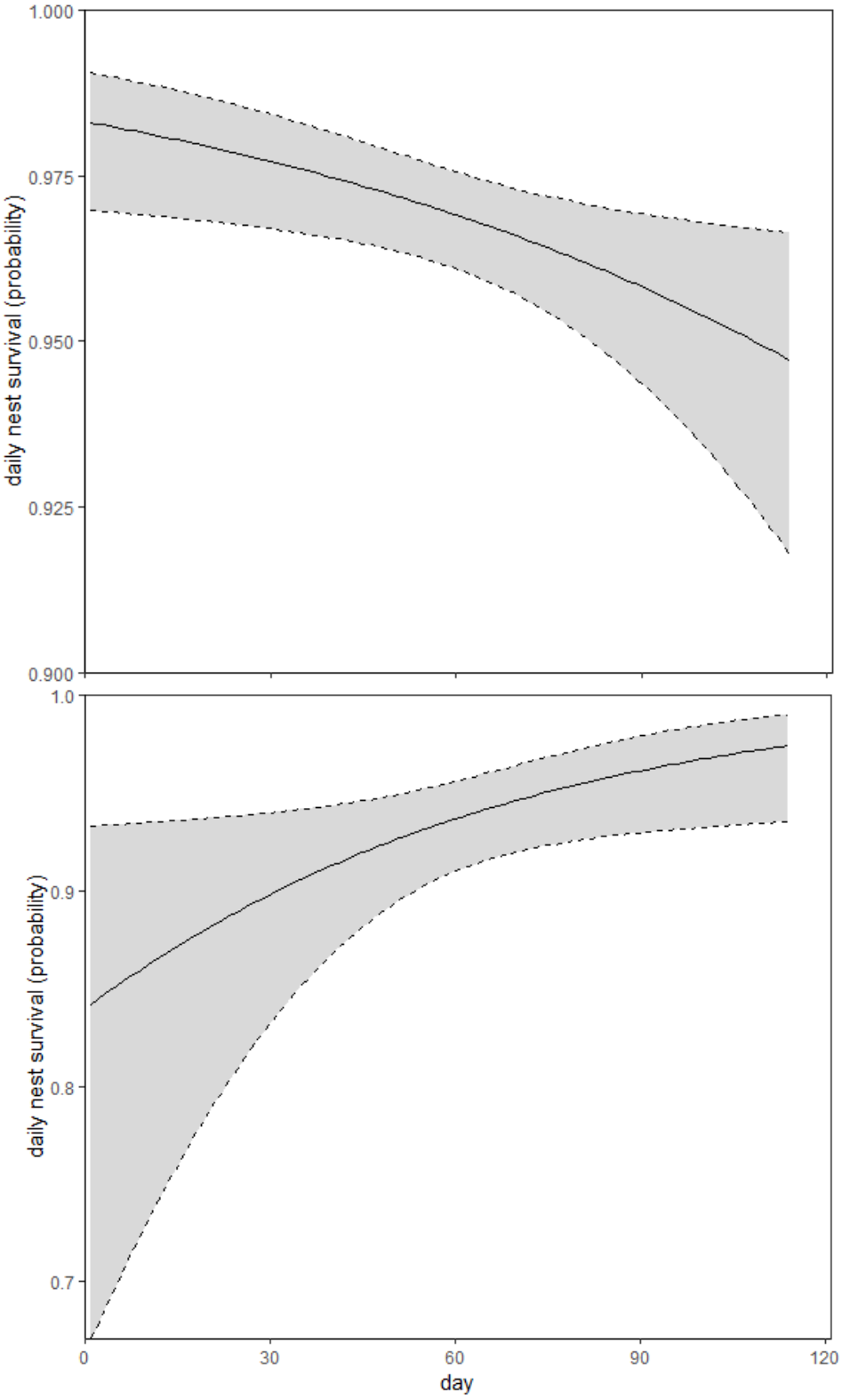

Figure 3.3 Daily nest survival of dome-nesting species (top) and cup-nesting species (bottom) over the course of the 2016 spring breeding season in the South-west Slopes bioregion, NSW. Probability refers to the likelihood of the nest surviving to the end of the study. Day 1 represents the first day of the study (the first day on which a nest could be discovered). Shaded areas indicate upper and lower 95% confidence intervals. Plot created using ggplot2 for R (Wickham 2016).

Dome-nesters frequently nested in kangaroo thorn (*Acacia paradoxa*), red box (*Eucalyptus polyanthemos*), Blakely's red gum, and *Phalaris aquatica* (an introduced grass species). Plant species used frequently by cup-nesters included Blakely's red gum, white box, and kangaroo thorn. Both cup-nesters and dome-nesters nested most often in trees. Dome-nesters also frequently nested in shrubs and woody debris. Cup-nesters rarely built nests in shrubs or woody debris.

We found that in linear sites that were oriented north-south, it was common for nests to be located on the eastern side of the site. This meant that nests were exposed to the warmth of the early morning sun but protected from overheating in the afternoon.

Model selection results

When analysing nest fate, we found that the null model was retained for every assemblage and species of interest, and in every iteration of our analyses (Appendix 3.2–3.4). That is, none of the predictors in our candidate models explained the variation in nest fate. We found that analysing daily nest survival produced more conclusive results. When all sites were included (plantings, remnants, large reference sites), the null model was again retained in every instance (Appendix 3.5). We found that candidate models performed better against the null model when large reference sites were excluded (Table 3.3). We did not find any conclusive results when analyses were restricted to restoration plantings (Appendix 3.6). Note that we had sufficient data to examine only dome-nesters and the superb fairy-wren in restoration plantings.

Table 3.3 Daily nest survival models for woodland birds in restoration plantings and similarly sized woodland remnants (excluding large reference sites). Models are ranked by Akaike's Information Criterion corrected for small sample sizes (AIC_c).

Dome-nesters	npar	AIC_c	ΔAIC_c	AIC_w	Deviance
Constant	1	343.94	0.00	0.73	341.93
TYPE + SIZE + SHAPE + DATE	5	346.53	2.59	0.20	336.49
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	348.82	4.88	0.06	338.78
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	352.06	8.12	0.01	337.99
Cup-nesters					
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	134.88	0.00	0.75	124.74
Constant	1	137.74	2.86	0.18	135.73
TYPE + SIZE + SHAPE + DATE	5	140.18	5.29	0.05	130.03
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	142.46	7.57	0.02	128.19
Superb fairywren					
Constant	1	180.65	0.00	0.80	178.64
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	184.06	3.41	0.14	173.98
TYPE + SIZE + SHAPE + DATE	5	186.51	5.86	0.04	176.44
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	189.24	8.59	0.01	175.10
Species of conservation concern					
TYPE + SIZE + SHAPE + DATE	5	122.98	0.00	0.73	112.89
Constant	1	125.54	2.56	0.20	123.53
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	127.89	4.91	0.06	117.79
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	133.66	10.68	0.00	119.49

Effects of patch type

Daily nest survival for species of conservation concern was higher in plantings than in similarly sized woodland remnants (Table 3.4). Patch type did not influence daily nest survival for any other groups of interest. We did not identify any effect of patch type on nest fate for woodland birds in our study (Appendix 3.2, 3.3).

Table 3.4 Parameter estimates for daily nest survival models computed by Program MARK for species of conservation concern in restoration plantings and similarly sized woodland remnants (n = 31, n_{ess} = 647).

Parameter	Estimate (SE)
Intercept	4.77 (0.69)
TYPE (remnant)	–1.56 (0.97)
SIZE	–0.97 (0.34)
SHAPE	–0.57 (0.29)
DATE	–0.02 (0.01)

Importance of patch attributes

For species of conservation concern, daily nest survival in plantings and similarly sized woodland remnants was better predicted by patch attributes than by nest-site or microhabitat variables (Table 3.3). This was the only instance in which patch attributes outperformed the null model. In addition to the aforementioned effect of patch type, we found that daily nest survival for species of conservation concern decreased with increasing patch size (Table 3.4). We also found a negative effect of linearity, with lower daily nest survival in more linear sites.

Daily nest survival for cup-nesting species in plantings and similarly sized remnants was best predicted by nest-site variables (Table 3.3). We found a negative effect of nest height – nests situated higher above the ground were associated with lower survival probabilities (Table 3.5). Effects estimates for other variables in the model had large standard errors, and were therefore not interpretable.

Table 3.5 Parameter estimates for daily nest survival modelled against nest-site variables for cup-nesting species in restoration plantings and similarly sized woodland remnants (n = 39, n_{ess} = 428).

Parameter	Estimate (SE)
Intercept	1.75 (0.53)
DIST_EDGE	0.18 (0.24)
CONCEALMENT	0.04 (0.23)
HEIGHT	–0.61 (0.28)
DATE	0.01 (0.01)

Microhabitat variables were of little importance in determining breeding success of woodland birds in our study (Table 3.3, Appendix 3.2–3.7). Likewise, the age of restoration plantings did not contribute to predicting either nest fate or daily nest survival (Appendix 3.4, 3.7).

Causes of nest failure

The primary cause of nest failure was predation, which we identified as the cause of 91% of failed nests. This did not differ significantly between plantings, remnants, or large reference sites. Most other nest failures were attributed to abandonment, usually during the egg stage.

Nest-predators

We monitored 85 nests with cameras, and analysed a total of 308,249 camera trap images. Predation events recorded during our study were most often perpetrated by generalist avian predators, including ravens, the pied currawong (*Strepera graculina*), and pied butcherbird (*Cracticus torquatus*) (Table 3.6). The next most common nest-predator (and the most damaging individual species) identified in restoration plantings and woodland remnants was the red fox (Table 3.6; Appendix 3.7). Foxes targeted nests close to the ground, including those of the superb fairywren (Table 3.6). We also recorded some unexpected nest-predators, including the white-browed babbler (*Pomatostomus superciliosus*) (see Belder 2018), and

common ringtail possum (*Pseudocheirus peregrinus*). The eastern brown snake was recorded as a nest-predator in a restoration planting on one occasion (Table 3.6).

Table 3.6 Nest-predators identified from camera trap imagery of 85 monitored woodland bird nests in the South-west Slopes bioregion, NSW. Note that it was not possible to distinguish between Australian raven (*Corvus coronoides*) and little raven (*C. mellori*) on camera trap imagery. Cattle and sheep are included as predators on the basis of camera trap imagery, but may have destroyed nests without consuming eggs or nestlings.

Common name	Species	Planting	Remnant	Reference	Total	Nest height (m)
*red fox	*Vulpes vulpes*	5	1	2	8	0–1.1
Australian/little raven	*Corvus* sp.	5		1	6	0.9–2.2
pied currawong	*Strepera graculina*	1		2	3	0.3–5.5
pied butcherbird	*Cracticus torquatus*		2		2	1.6–1.7
brown goshawk	*Accipiter fasciatus*	1			1	1.7
white-browed babbler	*Pomatostomus superciliosus*	1			1	0.4
Australian magpie	*Gymnorhina tibicen*			1	1	1.1
common brushtail possum	*Trichosurus vulpecula*			1	1	0.1
common ringtail possum	*Pseudocheirus peregrinus*			1	1	4.0
*cattle	*Bos taurus*		1		1	0.3
*sheep	*Ovus aries*			1	1	0.3
eastern brown snake	*Pseudonaja textilis*	1			1	0.2
eastern blue-tongue lizard	*Tiliqua scincoides*	1			1	0.2

*Introduced species

Discussion

Our results are empirical evidence that restoration plantings provide suitable breeding habitat for woodland birds, and may eclipse remnant patches in supporting successful breeding of woodland birds. We found that woodland birds bred at least as successfully in restoration plantings as they did in remnant woodland patches and large reference sites. Indeed, species of conservation concern were more likely to breed successfully in restoration plantings than in remnant woodland patches. Other notable findings included negative effects of both patch size and linearity on daily nest survival for species of conservation concern.

Nest survival as measured in our study was somewhat lower than expected, particularly for cup-nesting species (29.6% for cup-nesting species and 38.1% for dome-nesting species). Nest survival for Australian songbirds of the families included in our study average 42.2% for dome-nesting species and 37.7% for cup-nesting species (Remeš *et al.* 2012). This may indicate that habitat suitability of restoration plantings and remnant patches in our study region is lower for cup-nesters than it is for dome-nesters. Many cup-nesting species are perch-and-pounce ground-foraging species, including the willie wagtail and various robins (Petroicidae). Species in the latter family have been identified as susceptible to population decline, and careful management of the ground layer has been recommended to improve habitat suitability for these species (Recher *et al.* 2002; Antos and Bennett 2006; Montague-Drake *et al.* 2009).

A decline in breeding success over the course of the breeding season, as documented for the dome-nester assemblage, is consistent with patterns observed for many bird species worldwide (Arnold *et al.* 2004). The positive effect of date on DSR that we recorded for cup-nesters was unexpected. Potential explanations include more stable weather conditions later in the season, lower predation risk (particularly by avian predators), or changes in microhabitat variables such as grass cover over the course of the breeding season.

Belder *et al.* (2019) documented equivalent levels of breeding activity in restoration plantings and woodland remnants, including for species of conservation concern. Our findings regarding breeding success are quantitative evidence that restoration plantings provide valuable habitat in which threatened and declining bird species can persist and breed. They also potentially highlight a need to improve the quality of woodland remnants through restorative actions such as excluding stock or replanting the shrub layer. Some species of

conservation concern, such as the brown treecreeper (*Climacteris picumnus*), rely on habitat features that are present in woodland remnants but take decades to develop in restoration plantings (Vesk *et al.* 2008). It is for this reason that restoration plantings should be considered complementary to, and not a replacement for, remnant woodland (Cunningham *et al.* 2008; Lindenmayer *et al.* 2018*d*; Ikin *et al.* 2018).

Previous studies have documented a positive relationship between patch size and reproductive output in birds (e.g. Burke and Nol 2000; Zanette *et al.* 2000; Zanette and Jenkins 2000; Zanette 2001). This has led to the prevalent view that larger patches are more valuable for woodland birds in fragmented agricultural landscapes. However, Belder *et al.* (2019) found that breeding activity in the South-west Slopes bioregion decreased with increasing patch size. The results of the present study substantiate this finding. Previous research has described the value of small patches for sustaining wildlife populations (Tulloch *et al.* 2016; Lindenmayer 2019; Wintle *et al.* 2019). Our study provides direct evidence that woodland birds are able to breed successfully in small habitat patches. Possible reasons for greater success in small patches include lower abundances of predators and brood parasites in small patches, the dominance of edge-specialists and habitat generalists, and concentration effects (Belder *et al.* 2019).

While linear patches may provide suitable habitat for some species (as evidenced by our general finding of little influence of linearity on breeding success), our results indicate that species of conservation concern may benefit more from block-shaped sites. This may be one reason why linear sites have previously been found to contain a less diverse species assemblage than block-shaped sites (Kinross 2004; Lindenmayer *et al.* 2010, 2018*b*). This is

of interest for conservation planning, as it highlights the need to take into account the habitat requirements of different species and assemblages when designing revegetation programs.

The presence of nest height as an explanatory variable in top models for cup-nesters may be a reflection of the dominant predators in the study region – open cup-nests are frequently targeted by avian predators (Okada *et al.* 2019), which may more easily locate these nests higher up in the canopy. We did not find any evidence that the distance of a nest to the nearest patch edge influenced breeding success. This is suggestive of a lack of edge-effects, which have been thought to decrease the value of small and/or linear habitat patches for birds in fragmented agricultural landscapes (Ewers and Didham 2007; King *et al.* 2009). However, as discussed earlier, our results showed that species of conservation concern bred more successfully in sites of decreasing linearity. One potential explanation is that linear sites do not facilitate optimal central place foraging, since nesting birds must expend more energy traversing a linear home range than one that is more uniform in shape (Andersson 1978; Bovet and Benhamou 1991; Rosenberg and McKelvey 2016).

It is somewhat surprising that microhabitat variables and planting age contributed little to explaining breeding success in our study. Previous research has documented the influence of variables such as shrub cover and ground layer complexity on site occupancy by woodland birds (Seddon *et al.* 2003; Antos and Bennett 2006; Montague-Drake *et al.* 2009; Munro *et al.* 2011). Belder *et al.* (2019) also reported increased breeding activity of woodland birds in younger restoration plantings, which the authors postulated was due to a diversity of nest-site choices and foraging opportunities associated with the presence of an intact shrub layer. It is possible that microhabitat variables other than the ones included in this study may have had a greater influence on breeding success.

The high nest-predation rate we recorded during our study is not unprecedented (see Zanette and Jenkins 2000; Guppy *et al.* 2017), but it is nonetheless of concern for the persistence of woodland bird populations in our study region. Generalist avian predators, including corvids, are often considered among species that have benefited from land clearing and habitat fragmentation in agricultural landscapes worldwide (Andrén 1992; Ford *et al.* 2001; Fuller *et al.* 2005). Invasive predators, including foxes, also benefit from increasing agricultural land-use (Graham *et al.* 2012). Zanette and Jenkins (2000) suggest that decreasing forest cover at the landscape scale is a key factor that has led to increased incidence of nest-predation. Measuring landscape-scale vegetation cover was outside the scope of our study, but more than five million hectares of white box / yellow box / Blakely's red gum grassy woodland has been cleared since European settlement, and less than 10% of this ecological community remains across its historic range (Manning *et al.* 2011). The low levels of landscape vegetation cover in our study region may be a significant influence on woodland bird population dynamics.

Prior to commencing this study, we predicted that predatory bird species and the eastern brown snake would be the dominant nest-predators in our study sites. While avian predators such as ravens and butcherbirds were indeed responsible for the majority of predation events captured during our study, we also identified another common predator – the introduced red fox. The prevalence of foxes as nest-predators in restoration plantings is cause for concern, and may limit the habitat suitability of plantings for woodland birds that nest in the lower strata or on the ground. These include several threatened and declining species, such as the speckled warbler (*Pyrrholaemus sagittatus*).

Inferential limitations

Our study has revealed previously undocumented trends in woodland bird breeding success, and provided insights into the capacity of restoration plantings and small habitat patches to support woodland birds. We acknowledge a number of limitations pertaining to the present study, and communicate these here to assist with interpretation.

First, this study was conducted over a short duration. The first field season was a pilot study that enabled collection of nest fate data only, leaving one field season in which we could collect sufficiently detailed data to calculate daily nest survival. Caution is advised when extrapolating from studies of only a year duration (Maron *et al.* 2005). The field season on which a majority of the data in this paper are based coincided with a year of above average rainfall. Since the productivity of southern temperate woodlands is strongly linked to soil moisture (Watson 2011), it is possible that breeding success in our study region may ordinarily be lower than documented in our study.

Second, the presence of the noisy miner, a hyperaggressive native honeyeater, in fragmented agricultural landscapes is a key threatening process for many woodland bird species (Montague-Drake *et al.* 2011; Bennett *et al.* 2015; Maron *et al.* 2016). The noisy miner harasses small woodland birds, is a known nest-predator, and has been directly implicated in reduced breeding success of woodland bird species (Maron 2007; Maron *et al.* 2013; Bennett *et al.* 2015; Beggs *et al.* 2019). Our study was conducted in the absence of the noisy miner, but it is important to recognise that this species may be present and adversely affect breeding success of woodland birds in restoration plantings or other woodland patches.

Future research directions and management implications

We suggest that future research should focus on identifying the reasons for low breeding success in linear patches, particularly given the popularity of such configured habitat patches in agricultural landscapes. We also recommend more detailed studies on breeding success of individual species. In particular, it would be worth focusing on robins and other declining cup-nesting species to identify reasons for low nest survival and more frequent failure earlier in the breeding season (*sensu* Zanette *et al.* 2000). More extensive habitat surveys (as opposed to a focus on the immediate vicinity of the nest) might prove useful in assessing the influence of habitat variables on breeding success. In terms of management, we recommend controlling for foxes to maximise the likelihood that restoration plantings and other woodland patches support breeding populations of woodland birds.

There is still much to be learned about woodland bird population dynamics in restored landscapes, and in fragmented agricultural landscapes generally. We suggest that future studies on the responses of woodland birds (and other fauna) to conservation strategies move beyond pattern data and adopt more detailed, population-oriented approaches such as the one presented in our study. Future research should focus on aspects of habitat quality that are likely to influence population persistence, such as identifying the major threats to woodland bird breeding. We also suggest that future studies be undertaken over longer time periods, to capture inter-annual variation in breeding success and reproductive output. This is particularly relevant in large parts of Australia, where animal populations fluctuate in response to extreme interannual variations in climate and rainfall (Letnic and Dickman 2006; Burbidge and Fuller 2007). Basing management outcomes on multiple years of study is a crucial component of ongoing successful biodiversity conservation. It would be highly beneficial to include studies such as ours in long-term monitoring projects, so that community responses to environmental

change can be documented. However, we acknowledge that monitoring breeding success is labour-intensive, time-consuming, and costly.

With the emergence of new wildlife monitoring technologies, including improvements in camera trap technology, we are hopeful that nest-monitoring will become easier and therefore more commonplace in bird breeding studies. A camera trapping method and/or software that could accurately and reliably determine key events in the nesting cycle (completion of building, egg-laying, hatching, nest predation, fledging, and abandonment) could revolutionise our ability to assess breeding success in studies worldwide. This would facilitate the incorporation of breeding studies into long-term monitoring projects, and importantly, in projects that aim to evaluate the success of conservation strategies.

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Appendices

Appendix 3.1 Number of nests by site and species. Conservation status: LC = least concern, CC = conservation concern, V = Vulnerable. Taxonomy follows Gill and Donsker (2018).

Species		Nest type	Conservation status	Plantings	Remnants	Reference sites
crested pigeon	*Ocyphaps lophotes*	cup	LC		1	
peaceful dove	*Geopelia placida*	cup	LC		1	
superb fairywren[P]	*Malurus cyaneus*	dome	LC	56	13	17
speckled warbler[P]	*Pyrrholaemus sagittatus*	dome	CC			1
white-browed scrubwren[P]	*Sericornis frontalis*	dome	LC	1		
weebill[P]	*Smicrornis brevirostris*	dome	CC	6		1
western gerygone[P]	*Gerygone fusca*	dome	LC	3		3
brown thornbill	*Acanthiza pusilla*	dome	LC			2
buff-rumped thornbill	*Acanthiza reguloides*	dome	LC	3	3	4
yellow-rumped thornbill[P]	*Acanthiza chrysorrhoa*	dome	CC	21	3	3
striated thornbill[P]	*Acanthiza lineata*	dome	LC			1
black-faced cuckooshrike	*Coracina novaehollandiae*	cup	LC		1	
white-winged triller	*Lalage tricolor*	cup	CC	1		
varied sittella	*Daphoenositta chrysoptera*	cup	V			3
rufous whistler[P]	*Pachycephala rufiventris*	cup	LC	3	1	1
grey shrikethrush[P]	*Colluricincla harmonica*	cup	LC	10	2	1
willie wagtail[P]	*Rhipidura leucophrys*	cup	LC	11	18	4
grey fantail[P]	*Rhipidura albiscapa*	cup	LC	5	1	3
eastern yellow robin[P]	*Eopsaltria australis*	cup	LC			1
hooded robin[P]	*Melanodryas cucullata*	cup	V			1
jacky winter	*Microeca fascinans*	cup	CC		1	1
red-capped robin[P]	*Petroica goodenovii*	cup	CC			1
diamond firetail[P]	*Stagonopleura guttata*	dome	V	5		
red-browed finch[P]	*Neochmia temporalis*	dome	LC	1		

[P] Species identified as plantings specialists

Appendix 3.2 Mixed effects models for nest fate of birds breeding in all study sites (restoration plantings, remnants, and large reference sites), modelled against patch attributes, nest-site variables and microhabitat variables. Models are ranked by Akaike's Information Criterion corrected for small sample sizes (AIC_c).

Dome-nesters	df	log(*L*)	AIC_c	ΔAIC_c
Intercept only	2	-68.1	140.38	0.00
TYPE + DATE	4	-67.1	144.84	4.46
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-66.4	150.35	9.97
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-66.4	152.67	12.29
Cup-nesters				
Intercept only	2	-28.1	60.43	0.00
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-21.2	64.59	4.16
TYPE + DATE	4	-26.7	64.74	4.31
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-24.8	69.00	8.57
Superb fairywren				
Intercept only	2	-29.0	72.18	0.00
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-26.1	77.77	5.59
TYPE + DATE	4	-27.7	78.26	6.08
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-24.6	81.75	9.57
Willie wagtail				
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	0.0	25.00	0.00
Intercept only	2	-10.4	26.53	1.53
TYPE + DATE	4	-5.2	32.98	7.98
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-5.4	40.75	15.75
Species of conservation concern (dome-nesters)				
Intercept only	2	-23.4	51.17	0.00
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-13.9	53.09	1.92
TYPE + DATE	4	-22.1	56.34	5.17
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-20.1	61.84	10.67

Appendix 3.3 Mixed effects models for nest fate of birds breeding in restoration plantings and similarly sized woodland remnants, modelled against patch attributes, nest-site variables and microhabitat variables. Models are ranked by Akaike's Information Criterion corrected for small sample sizes (AIC_c).

Dome-nesters	df	log(*L*)	AIC_c	ΔAIC_c
Intercept only	2	-54.0	112.23	0.00
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-52.7	123.30	11.07
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-51.8	123.87	11.64
TYPE + SIZE + SHAPE + TYPE:SIZE + TYPE:SHAPE + SIZE:SHAPE + DATE	9	-52.5	125.30	13.07
Cup-nesters				
Intercept only	2	-22.5	49.31	0.00
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-16.6	57.16	7.85
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-19.0	58.72	9.41
TYPE + SIZE + SHAPE + TYPE:SIZE + TYPE:SHAPE + SIZE:SHAPE + DATE	8	-19.1	62.30	12.99
Superb fairywren				
Intercept only	2	-34.0	62.18	0.00
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-29.4	72.13	9.95
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-29.9	72.21	10.03
TYPE + SIZE + SHAPE + TYPE:SIZE + TYPE:SHAPE + SIZE:SHAPE + DATE	4	-33.5	78.40	16.22
Willie wagtail				
Intercept only	2	-11.0	25.49	0.00
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	0.0	27.08	1.59
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-5.4	43.75	18.26
TYPE + SIZE + SHAPE + TYPE:SIZE + TYPE:SHAPE + SIZE:SHAPE + DATE	4	-9.9	43.39	17.90
Species of conservation concern (dome-nesters)				
Intercept only	2	-21.4	47.13	0.00
TYPE + SIZE + SHAPE + TYPE:SIZE + TYPE:SHAPE + SIZE:SHAPE + DATE	9	-17.1	52.90	5.77
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-13.7	53.56	6.43
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-19.3	60.84	13.71

Appendix 3.4 Mixed effects models for nest fate of birds breeding in restoration plantings, modelled against patch attributes, nest-site and microhabitat variables, and planting age. Models are ranked by Akaike's Information Criterion corrected for small sample sizes (AIC_c).

Dome-nesters	df	log(*L*)	AIC_c	ΔAIC_c
Intercept only	2	-46.3	96.76	0.00
AGE + DATE	4	-45.7	100.00	3.24
SIZE + SHAPE + SIZE:SHAPE + DATE	6	-45.4	104.07	7.31
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-43.5	107.82	11.06
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-45.2	108.62	11.86
Cup-nesters				
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	0.0	28.00	0.00
AGE + DATE	4	-9.3	29.02	1.02
Intercept only	2	-12.6	29.79	1.79
SIZE + SHAPE + SIZE:SHAPE + DATE	6	-8.8	35.52	7.52
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-8.9	45.87	17.87
Superb fairywren				
Intercept only	2	-24.0	52.25	0.00
AGE + DATE	4	-22.7	54.61	2.36
SIZE + SHAPE + SIZE:SHAPE + DATE	6	-23.5	61.72	9.47
LEAF LITTER + BARE GROUND + WOODY DEBRIS + GRASS + SHRUB COVER + DATE	8	-21.7	64.47	12.22
DIST_EDGE + HEIGHT + CONCEALMENT + DIST_EDGE:HEIGHT + DIST_EDGE:CONCEALMENT + HEIGHT:CONCEALMENT + DATE	9	-20.6	65.92	13.67

Appendix 3.5 Daily nest survival models for woodland birds in all study sites (restoration plantings, remnants, and large reference sites). Models are ranked by Akaike's Information Criterion corrected for small sample sizes (AIC_c).

Dome-nesters	npar	AIC_c	ΔAIC_c	AIC_w	Deviance
TYPE + DATE	4	432.59	0.00	0.35	424.57
Constant	1	432.70	0.12	0.33	430.70
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	433.14	0.56	0.27	423.12
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	436.54	3.96	0.05	422.49
Cup-nesters					
Constant	1	182.25	0.00	0.76	180.24
TYPE + DATE	4	185.70	3.45	0.13	177.64
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	186.53	4.28	0.09	176.43
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	189.68	7.43	0.02	175.49
Superb fairywren					
Constant	1	228.14	0.00	0.79	226.13
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	232.10	3.96	0.11	222.04
TYPE + DATE	4	232.95	4.81	0.07	224.91
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	234.34	6.20	0.04	220.23
Species of conservation concern					
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	137.33	0.00	0.60	127.25
Constant	1	138.50	1.17	0.36	136.49
TYPE + DATE	4	142.36	5.03	0.05	134.31
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	145.13	7.80	0.01	130.97

Appendix 3.6 Daily nest survival models for woodland birds in restoration plantings. Models are ranked by Akaike's Information Criterion corrected for small sample sizes (AIC_c).

Dome-nesters	npar	AIC_c	ΔAIC_c	AIC_w	Deviance
Constant	1	283.44	0.00	0.57	281.44
SIZE + SHAPE + DATE	4	285.21	1.77	0.24	277.18
AGE + DATE	3	286.08	2.64	0.15	280.06
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	289.65	6.21	0.03	279.61
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	291.06	7.61	0.01	276.98
Superb fairywren					
Constant	1	143.57	0.00	0.52	141.56
AGE + DATE	3	144.44	0.87	0.34	138.40
DIST_EDGE + CONCEALMENT + HEIGHT + DATE	5	147.49	3.92	0.07	137.40
SIZE + SHAPE + DATE	4	147.73	4.17	0.06	139.67
GRASS + BARE GROUND + LEAF LITTER + WOODY DEBRIS + SHRUBS + DATE	7	152.60	9.04	0.01	138.43

Appendix 3.7 A red fox (*Vulpes vulpes*), bottom right, pouncing on the nest of a superb fairywren (*Malurus cyaneus*), in the South-west Slopes bioregion, NSW. Infrared imagery captured by a Bushnell HD Trophy Cam. Photo: Donna Belder.

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CHAPTER 4

Ongoing declines of woodland birds: are restoration plantings making a difference?

A colour-banded male superb fairywren (*Malurus cyaneus*) in a restoration planting in the South-west Slopes bioregion, NSW. Photo: Madaline Hill.

Belder, D. J., Pierson, J. C., Rudder, A. C., and Lindenmayer, D. B. (2020). Ongoing declines of woodland birds: are restoration plantings making a difference? *Ecological Applications* (under review).

Abstract

Woodland birds are a species assemblage of conservation concern, and their persistence in fragmented agricultural landscapes is dependent on both the preservation of existing woodland remnants and the implementation of restoration plantings. However, little is known about the habitat-use and persistence of birds in fragmented agricultural landscapes. We present a detailed, population-oriented study of woodland birds in temperate eucalypt woodland restoration plantings and remnant woodland patches in the South-west Slopes bioregion of New South Wales, Australia. First, we undertook a three-year mark-recapture project to assess annual survival and site fidelity in restoration plantings and woodland remnants. We supplemented our recapture efforts with resightings of colour-banded individuals. Second, we tracked individual birds of two species – superb fairywren (*Malurus cyaneus*) and willie wagtail (*Rhipidura leucophrys*) – and documented snapshots of their home ranges and movement patterns during the breeding season. Annual survival in the woodland bird assemblage was lower than expected (51%). Home ranges of the superb fairywren were positively correlated with patch size, and were constrained by patch edges in linear sites. Superb fairywrens and willie wagtails were more likely to travel longer distances between substrates while foraging in linear sites. Willie wagtails engaged in significant gap-crossing (up to 400 m) between adjacent habitat patches. Our findings indicate that 1) patch isolation and certain patch configurations place resident birds at an energetic disadvantage, and 2) in our study area, woodland bird populations are continuing to decline. We recommend landscape-scale habitat restoration programs aim to address ongoing population declines. Studies such as ours conducted over longer time periods would provide a deeper understanding of habitat-use and population processes of woodland birds in fragmented agricultural landscapes.

Key words: ringing, mark-recapture, population dynamics, territory, animal movement

Introduction

Habitat loss, fragmentation, and degradation resulting from agricultural expansion are leading causes of biodiversity declines worldwide (Newbold *et al.* 2015; Molotoks *et al.* 2018). The current rate of global biodiversity loss has been described as "catastrophic" (Driscoll *et al.* 2018), with experts fearing we have entered a sixth global mass extinction event (Barnosky *et al.* 2011; Ceballos *et al.* 2015). As such, there has been an increase in ecological restoration efforts to combat habitat loss and fragmentation (Menz *et al.* 2013; Barral *et al.* 2015). An example is the establishment of "restoration plantings" in agricultural landscapes: patches of replanted native vegetation in areas that have been previously cleared for agriculture.

In southern Australia, the recovery of woodland birds is a common objective of restoration plantings, as this group has suffered substantial and ongoing population declines due to habitat loss and fragmentation (Freudenberger 2001; Rayner *et al.* 2014). Several studies have identified a number of woodland bird species that preferentially occupy restoration plantings over woodland remnants (Barrett *et al.* 2008; Cunningham *et al.* 2008; Lindenmayer *et al.* 2010*b*). However, the vast majority of previous studies have used pattern data, such as presence and abundance, to draw conclusions about the suitability of habitat for woodland birds (Belder *et al.* 2018). Little is known about the long-term population responses and habitat-use of woodland birds in restoration plantings, and in fragmented agricultural landscapes more generally.

An understanding of population processes, including survival and site fidelity, is crucial for effective management of woodland bird populations (McKibbin and Bishop 2012; Belder *et al.* 2018). Survival is a key indicator of the extent to which restoration plantings and woodland remnants are providing suitable habitat for woodland birds. If survival is low,

habitat patches may be acting as population sinks or ecological traps (Dias 1996; Battin 2004). Site fidelity also provides insights into the relative importance of individual habitat patches for animal populations (McKibbin and Bishop 2012; Meager *et al.* 2018). If site fidelity is high, managers might allocate resources to maintain or improve habitat quality in targeted areas (Lehnen and Rodewald 2009). However, if site fidelity is low, a landscape-scale approach to habitat restoration and management may be needed (Schlossberg 2009).

In conjunction with demographic parameters such as survival and site fidelity, examining home ranges and movement patterns of individuals can provide insights into habitat quality and resource use. For example, the formation of smaller home ranges within a contiguous area typically indicates higher quality habitat (Ford 1983). Restoration plantings present an interesting paradox – many bird species will preferentially occupy plantings, but many plantings are narrow, linear windbreaks that are not conducive to optimal foraging according to established theories (Ford 1983; Stephens 2008). This may disproportionately affect species that are not highly mobile or not willing to cross habitat gaps (Van Houtan *et al.* 2007; Lees and Peres 2009). Assessing the effects of patch geometry and configuration on home ranges and movements of woodland birds may further assist with conservation planning and management.

Research objectives

The primary aim of this paper was to investigate whether restoration plantings are providing quality habitat to support woodland bird populations in fragmented agricultural landscapes. We used two indicators of habitat quality: annual survival/site fidelity, and home range attributes (size, shape).

We first sought to quantify annual survival and site fidelity in our study region. We predicted that survival and site fidelity in restoration plantings would be high, as we have 20 years of survey data in our study region suggesting that woodland birds readily occupy restoration plantings (Lindenmayer *et al.* 2018). We expected that birds in woodland remnants would show even higher survival and site fidelity, as these patches are typically considered to support a wider range of species than restoration plantings (Cunningham *et al.* 2008; Ikin *et al.* 2018). In addition, most of the species chosen for our study are sedentary, and long-distance movements are not commonly reported (Australian Bird and Bat Banding Scheme 2016). In general, we expected high survival estimates. This is because many species of Australian birds are long-lived compared to their similarly sized northern hemisphere counterparts – lifespans of 20+ years have been recorded for many species (Yom-Tov *et al.* 1992; Australian Bird and Bat Banding Scheme 2016). A high survival rate should be expected for Australian woodland bird species if the habitat is suitable.

In addition to quantifying survival and site fidelity, we posed the following specific research questions:

Question 1. Do patch attributes (type, size, shape) influence survival and site fidelity of woodland birds in fragmented agricultural landscapes?

We expected survival and site fidelity to be higher in block-shaped sites than in linear sites. This is because previous studies have indicated that block-shaped sites support a greater diversity and abundance of woodland bird species than linear sites (Lindenmayer *et al.* 2007). Previous research has also linked patch size to habitat quality in fragmented agricultural landscapes, with many studies recommending that restoration plantings should be as large as possible to maximise biodiversity benefits (Freudenberger *et al.* 2004; Lindenmayer *et al.*

2010*b*; Cunningham *et al.* 2015). We therefore predicted that survival and site fidelity would be higher in larger sites than in smaller sites.

Question 2. How do patch attributes influence the movement patterns of birds, including the size and shape of home ranges?

We predicted that home ranges would be smaller in reference sites than in restoration plantings and similarly sized woodland remnants. This is because reference areas present a contiguous area of suitable habitat in which birds can establish a territory. When resources are patchily distributed, as they are in fragmented habitats, birds may need to move greater distances through the landscape to obtain sufficient resources (Ford 1983; Hinsley *et al.* 2008).

We expected the home ranges of individuals in block-shaped sites to be more compact and rounded than those in linear sites. This is because optimal foraging theory favours the formation of a more uniform home range shape (Andersson 1978; Dill 1978). This is especially relevant during the breeding season, when optimal central place foraging is a key determinant of home range size and shape (Andersson 1981; Kacelnik 1984; Rosenberg and McKelvey 2016). Optimal central place foraging is foraging away from a key central point, such as the prime habitat within a defended territory, or an active nest (Stephens 2008).

We predicted that birds in linear sites would travel longer average distances between distinct substrates within their territories. We also expected birds in smaller and more linear sites to engage in more gap-crossing behaviour than those in larger and more block-shaped sites.

Methods

Study area

We conducted our study in the South-west Slopes bioregion of New South Wales, Australia. It is the most cleared and fragmented bioregion in NSW, with at least 85% of the native temperate woodland vegetation having been cleared for agriculture since European settlement (Lindenmayer *et al.* 2010*a*; Benson 2013). Remnant patches persist as highly fragmented patches among a matrix of pasture and cropland, some as small as a single, isolated "paddock tree" (Lindenmayer *et al.* 2010*a*; Lindenmayer 2017). The majority of remnant vegetation persists on private land, and larger areas are predominantly on unproductive land such as rocky ridgetops. The dominant ecological community is white box (*Eucalyptus albens*) / yellow box (*E. melliodora*) / Blakely's red gum (*E. blakelyi*) grassy woodland, which is federally listed as critically endangered (Department of the Environment and Energy 2018). Patches of red stringybark (*E. macrorhyncha*) woodland and mugga ironbark (*E. sideroxylon*) woodland are also present. To address habitat loss, restoration plantings have been implemented on farms throughout the South-west Slopes bioregion (Lindenmayer *et al.* 2013). Woodland habitat in the study region is crucial for the persistence of several threatened bird species, including the critically endangered Regent Honeyeater (*Anthochaera phrygia*) and Swift Parrot (*Lathamus discolor*).

The South-west Slopes has a mild temperate climate, with warm to hot summers (January mean maximum temperature 31.5°C) and cool to cold winters (July mean maximum temperature 11.9°C) (Bureau of Meteorology 2019). Mean annual rainfall is 572 mm, and rainfall is distributed fairly evenly throughout the year (Bureau of Meteorology 2019). Rainfall in the study area was above average during the first two years of this study, and below average during the third year (Bureau of Meteorology 2019).

Study sites

We used long-term bird survey data (see Lindenmayer *et al.* 2010*b*) to select a subset of sites in which to conduct our study (Figure 4.1). We selected 12 restoration plantings and six similarly sized woodland remnants of varying size (1.3-7.7 ha) and shape (linear vs. block-shaped; calculated numerically as perimeter/width). We additionally chose three large (>47 ha) remnant patches of woodland to serve as reference sites, representing good quality woodland in the study area. Site attributes are detailed in Appendix 2.1.

We chose sites with low abundances of the noisy miner (*Manorina melanocephala*), a hyperaggressive native honeyeater that is known to exclude small woodland birds (Maron *et al.* 2013; Bennett *et al.* 2015; Beggs *et al.* 2019). While we acknowledge the significance of the noisy miner's influence on woodland birds, minimising its presence in our study sites enabled us to focus on our questions of interest. Our sites were separated geographically by a minimum of 500 m to promote spatial independence.

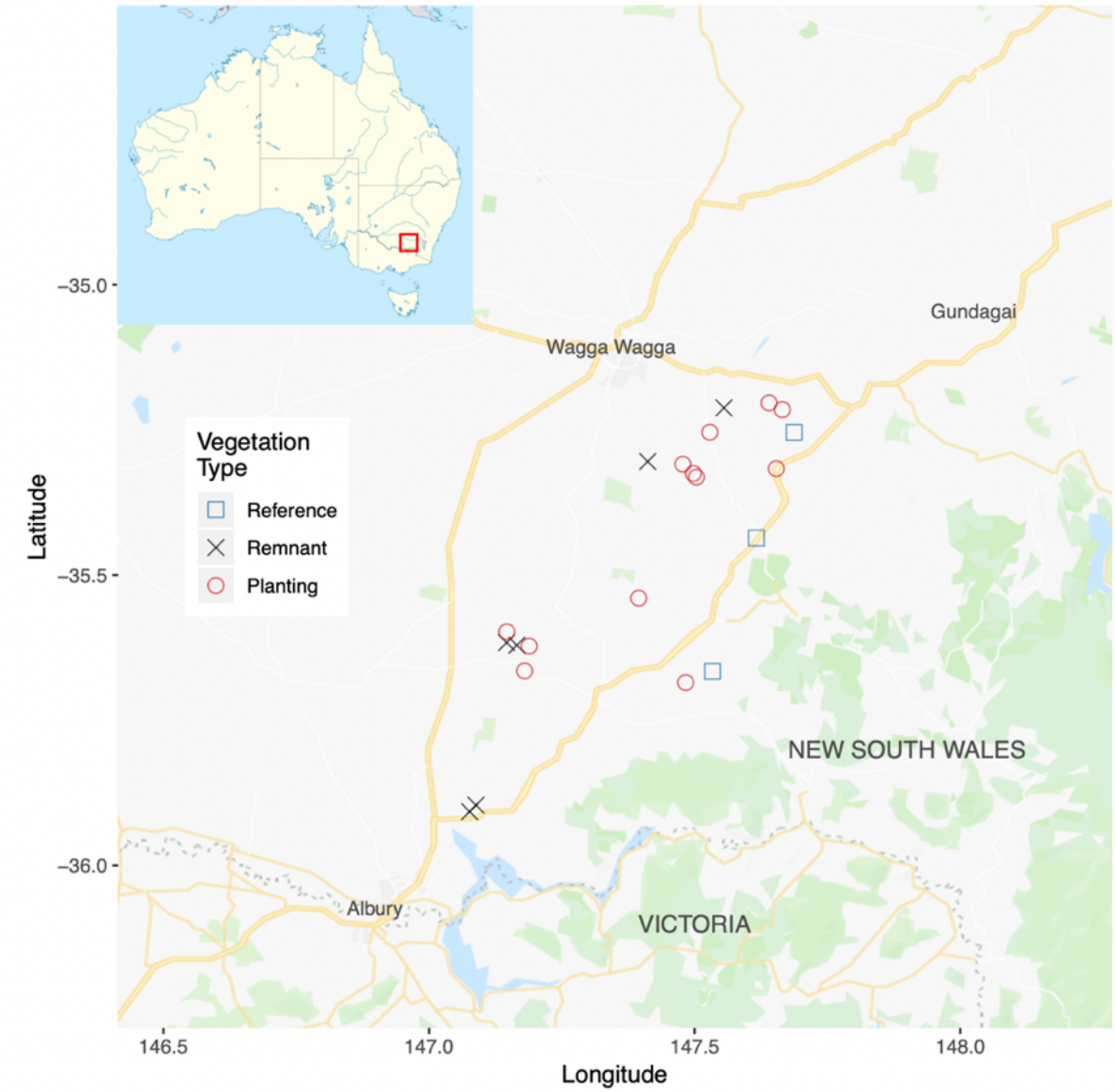

Figure 4.1 Location of study sites in the South-west Slopes bioregion of New South Wales, Australia. Map created using ggmap for R (Kahle and Wickham 2013).

Restoration plantings were aged between 12 and 25 years, and were characterised by a *Eucalyptus* overstorey and an understorey of predominantly *Acacia* shrubs. Trees and shrubs were planted for ecological purposes, and were usually fenced for protection from grazing by livestock (Appendix 2.1). Remnant woodland patches typically constituted a *Eucalyptus* overstorey plus an *Acacia*-dominated understorey of varying density. Remnant sites tended to contain more coarse woody debris (fallen branches and trees) than restoration plantings. All sites featured a ground layer that was usually dominated by exotic pasture grasses, with

various amounts of leaf litter, native forbs, native grasses, weeds, bare ground, moss/lichens, rocks, and coarse woody debris.

Bird banding and recaptures

We undertook initial banding of woodland birds in July-October of 2015. Subsequent recapture and banding efforts took place in June-August of 2016 and 2017. We conducted an approximately equal number of net hours in each study site, with net sites selected based on suitable habitat features and observed passage routes to maximise capture rates. Once selected, we used the same net sites consistently throughout the three years of the study, except when vegetation growth or tree collapse prevented the use of existing net sites. In these cases, we moved nets to suitable locations as close as possible to the original net sites. We used two nets that were 6 m in length, and four nets that were 9 m in length. We banded over two consecutive days in each study site. On the first day, we operated nets for approximately four hours pre-sunset. On the subsequent day, we opened nets from half an hour pre-dawn to approximately four hours post-sunrise. Inclement weather sometimes required us to close nets, and occasionally prevented us from banding on consecutive days in our study sites. In these instances, we resumed banding at the earliest opportunity once the weather had cleared.

We banded birds with a standard metal band (aluminium or aluminium alloy) provided by the Australian Bird and Bat Banding Scheme. Ten species also received a combination of coloured plastic and/or metal leg bands. These were the superb fairywren (*Malurus cyaneus*), yellow-rumped thornbill (*Acanthiza chrysorrhoa*), buff-rumped thornbill (*A. reguloides*), grey shrikethrush (*Colluricincla harmonica*), rufous whistler (*Pachycephala rufiventris*), red-capped robin (*Petroica goodenovii*), willie wagtail (*Rhipidura leucophrys*), white-browed babbler (*Pomatostomus superciliosus*), speckled warbler (*Pyrrholaemus sagittatus*), and

diamond firetail (*Stagonopleura guttata*). We released birds within 300 m of their initial capture site. Any juveniles captured were released as close as possible to their initial capture site.

For the purposes of our study, the "woodland assemblage" included all species that were captured in our study sites (Appendix 4.1).

Resightings

In the breeding season of 2015 (September-December), we recorded incidental sightings of colour-banded birds. In 2016, we undertook surveys to record sightings of colour-banded individuals. We undertook random area searches in each study site, with the length of time designated per unit area. We surveyed small sites (1.4 ha search area) for one hour, and large sites (3 ha search area) for two hours. A skilled observer searched a patch for woodland birds, and recorded colour-band combinations and GPS locations of colour-banded individuals when sighted. Sites were surveyed once per month from September to November. We also recorded incidental sightings of colour-banded individuals while visiting study sites throughout the breeding season.

The seasonally migratory rufous whistler had not yet returned to the study region when recapture efforts took place in 2016 and 2017. We resighted colour-banded individuals incidentally in 2015, and during thorough site surveys in 2016 (see Belder *et al.* 2019). In 2017, we undertook targeted surveys to identify whether colour-banded rufous whistlers had returned to their territories. We visited known territories of colour-banded males in October 2017 and used call-playback for up to 5 minutes to elicit a territorial response. If a whistler

responded, we immediately ceased playback and followed the bird until we could ascertain its band status/combination.

Home range tracking

We chose two target species for home range tracking – the superb fairywren, and the willie wagtail. Both species are relatively common in habitat patches in agricultural landscapes, and can use the matrix to some extent (usually foraging at the edges of patches). However, the two species differ in their movement patterns and habitat preferences (Menkhorst *et al.* 2017). Additional reasons for choosing these two species included their bold nature, and their ease of detection (facilitating tracking of individuals).

Superb fairywrens were fitted with coloured leg bands for ease of tracking and identification. For consistency, we tracked only male fairywrens. Not all tracked willie wagtails had been banded. However, the low density of willie wagtails in the study sites, their preference for open habitats, and the presence of individually distinct plumage or feather moult characteristics enabled us to reliably track individual birds over the course of several hours.

We selected birds to track at random. Upon arriving at a study site, an observer would locate an individual (typically the first bird encountered) on which to focus. Based on a pilot study in 2015, we determined that a minimum of 30 home range points was required to map a representative snapshot of the home range of a male superb fairywren or a willie wagtail. We therefore followed birds until at least 30 points had been recorded. We followed birds for a maximum of four hours.

We used a handheld Garmin eTrex GPS device, accurate to the nearest 3 m, to record home range data. Observers typically watched birds from a distance of around 20 m, using a pair of handheld binoculars, so as to not disrupt or influence the behaviour or movements of the birds. Observers also waited until a bird had moved away from each distinct substrate before approaching to record the GPS location.

Statistical analyses

Survival and site fidelity

To analyse recapture and resighting data, we used Program MARK (White and Burnham 1999) via the R package "RMark" (Laake 2003) in R version 3.5.2 (R Core Team 2019). We used the Cormack-Jolly-Seber (CJS) model (Lebreton *et al.* 1992) to estimate apparent survival (ϕ) for the entire assemblage. For colour-banded birds, and for the superb fairywren, we also used the Barker model (Barker 1997) to estimate survival and site fidelity. This model allows the inclusion of *ad hoc* resightings of colour-marked individuals, separates permanent emigration from true mortality, and relaxes the assumption of no permanent emigration. These modifications permit the calculation of an apparent survival estimate (S) that approaches true survival (Schwarzer *et al.* 2012; Barbour *et al.* 2013). The standard equation presented by Model parameters as described by Schwarzer *et al.* (2012) are as follows:

S_i survival

p_i recapture probability (the probability that an animal at risk of capture at time i is captured)

r_i the probability that an animal dies, is found and reported dead between time i and time $i + 1$

R_i resighting probability (the probability that an animal is resighted between time i and $i + 1$)

R_i' the probability that an animal dies between time i and $i + 1$, but is resighted alive before it dies

F_i fidelity (the probability that an animal at risk of capture at time i is at risk of capture at $i + 1$)

F_i' return (the probability that an animal not at risk of capture at time i is at risk of capture at $i + 1$)

As we recovered only one dead bird during our study, model outputs for r_i and R_i' were not interpretable. We therefore set these parameters to zero in all models.

We tested candidate models against a null model that assumed all parameters were constant except those fixed at zero (Table 4.1). Variables of interest were site type, size, and shape. To address overdispersion, we ranked candidate models by $QAIC_c$ adjusted by the variance inflation factor ($\hat{c}$) of the global model. We considered models with $\Delta QAIC_c \leq 2$ to be top-ranked models. We obtained two parameter estimates for each subset: the estimate from the top-ranked model, and model-averaged estimates from the top-ranked models. We used the package "MuMIn" (Bartoń 2018) in conjunction with "RMark" to conduct model averaging.

Table 4.1 Parameters modelled with CJS and Barker survival models. ϕ represents survival estimate from CJS models. S represents survival estimate from Barker models. Table adapted from Kauffman *et al.* (2003).

Model description	Parameters
Basic CJS model	ϕ(.) p(.)
Survival	
Constant	ϕ(.)
Site type	ϕ(type)
Site size	ϕ(size)
Site shape	ϕ(shape)
Recapture	
Constant	p(.)
Site type	p(type)
Site size	p(size)
Site shape	p(shape)
Basic Barker model	S(.) p(.) r(.) R(.) R'(.) F(.) F'(.)
Survival	
Constant	S(.)
Site type	S(type)
Site size	S(size)
Site shape	S(shape)
Recapture	
Constant	p(.)
Site type	p(type)
Site size	p(size)
Site shape	p(shape)
Recovery	
No dead recoveries	Fixed at zero
Resighting	
Constant	R(.)
Resighting'	
No dead recoveries	Fixed at zero
Fidelity	
Constant	F(.)
Site type	F(type)
Site size	F(size)
Site shape	F(shape)
Return parameter	
Constant	F'(.)

Home ranges and movement analyses

We used ArcMap Desktop version 10.6.1 (ESRI 2018) to plot GPS coordinates of home ranges. We used the Minimum Bounding Geometry function to calculate 100% Minimum Convex Polygons (MCPs) for all home ranges. We also calculated the perimeter and width of

each home range. Additionally, we calculated the distance moved by individual birds between distinct substrates (GPS points) during a tracking period, and documented "gap-crossing" behaviour. We defined "substrates" as distinct units of vegetation (individual trees, shrubs), woody debris (logs, fallen branches), artificial substrates (fence wires, posts), and patches of ground on which a bird perched or foraged. We defined gaps as spaces between patches of native vegetation. A "patch" of native vegetation may comprise a single paddock tree.

We used linear regression to model home range size and shape against patch attributes (type, size, shape, planting age) (Table 4.2). Home range size was log-transformed to improve model fit. We also modelled the average distance moved by individual birds between distinct substrates against patch attributes. Additionally, we investigated whether patch attributes influenced the occurrence of long-distance movements between substrates (movements approximately four times the average distance moved; >50 m in superb fairywrens and >100 m in willie wagtails). For the latter analyses, we used linear mixed effects regression models with individual bird as a random effect. We used a reciprocal transformation on movement distance to improve model fit. We used the R package "lme4" (Bates *et al.* 2015) to conduct linear mixed effects regression.

Table 4.2 Linear regression and linear mixed effects regression model parameters.

Variable name	Variable type	Description
TERRSIZE	Response	Home range size (ha)
TERRSHAPE	Response	Measure of home range shape, calculated as perimeter/width (m)
DIST	Response	Distance moved between substrates (m)
TYPE	Predictor	Patch type (planting, remnant, reference)
SIZE	Predictor	Patch size (ha)
SHAPE	Predictor	Measure of patch shape, calculated as perimeter/width (m)
AGE	Predictor	Age of planting at the commencement of the study (years)
BIRD	Random effect	Identity of tracked bird

Results

General

We banded 1261 woodland birds, comprising 38 species, over the duration of the study. The most commonly caught species were the superb fairywren, yellow-rumped thornbill, yellow thornbill, red-browed finch, and flame robin (Appendix 4.1). Our banding and recapture data are summarised in full in Appendix 4.1 and Appendix 4.2.

Survival and site fidelity

Of the 943 individuals banded during the first two years of the study, 386, or 40.9%, were recaptured or resighted at least once over the duration of the study. We achieved recapture rates of 18.3% and 6.8% in 2016 and 2017, respectively.

Annual survival estimates for the woodland assemblage according to CJS were 51% (SE=8%) (Table 4.3, Table 4.4). Survival estimates from Barker models for colour-banded birds and the superb fairywren were higher than estimates from CJS models (Table 4.3, Table 4.4). Barker survival estimates were 55 (SE=2%) for colour-banded birds, and 55 (SE=3%) for the superb fairywren (Table 4.3, Table 4.4). CJS and Barker estimates changed only slightly for each subset when reference sites were excluded from analyses. Although site attributes (type, size, shape) appeared in several top-ranked candidate models (Appendix 4.3–4.8), the null model was the best-fitted model in every instance.

Table 4.3 Survival estimates for birds in restoration plantings, similarly sized woodland remnants, and large reference sites. Estimates are taken from the best-fitted models for each subset. The null model was the best-fitted model in all instances.

Parameter	Estimate	SE	lcl (95%)	ucl (95%)
Woodland assemblage				
ϕ(.)	0.5109	0.0831	0.3524	0.6673
Colour-banded birds				
ϕ(.)	0.4672	0.0792	0.3197	0.6206
S(.)	0.5538	0.0233	0.5078	0.5989
Superb fairywren (*Malurus cyaneus*)				
ϕ(.)	0.4007	0.0729	0.2693	0.5482
S(.)	0.5491	0.0313	0.4874	0.6094

Table 4.4 Survival estimates for birds in restoration plantings and similarly sized woodland remnants (excluding large reference sites). Estimates from the best-fitted model and model-averaged estimates are provided for each subset. The null model was the best-fitted model in all instances.

Parameter	Estimate	SE	lcl (95%)	ucl (95%)
Woodland assemblage				
Best-fitted model				
ϕ(.)	0.5156	0.0880	0.3480	0.6798
Model average				
ϕ	0.5139	0.0877	0.3469	0.6778
Colour-banded birds				
Best-fitted model				
ϕ(.)	0.4780	0.0847	0.3201	0.6404
S(.)	0.5451	0.0241	0.4975	0.5919
Model average				
ϕ	0.4767	0.0846	0.3190	0.6392
S	0.5450	0.0242	0.4974	0.5919
Superb fairywren (*Malurus cyaneus*)				
Best-fitted model				
ϕ(.)	0.4190	0.0792	0.2759	0.5772
S(.)	0.5459	0.0321	0.4826	0.6077
Model average				
ϕ	0.4163	0.0788	0.2741	0.5739
S	0.5460	0.0323	0.4823	0.6083

Site fidelity estimates from Barker models for both colour-banded birds and the superb fairywren had very large confidence intervals, and were therefore not interpretable. However,

a noteworthy finding was that a large proportion of banded male rufous whistlers returned to the study sites. Of the nine males banded in 2015, seven returned in 2016 to defend the territories in which they were banded, and five returned again in 2017. Of the birds present in three consecutive years, four were in plantings, and one was in a woodland remnant. Interestingly, all of these plantings were linear in shape.

Home ranges

General

We mapped home ranges for 45 superb fairywrens and 30 willie wagtails (Table 4.5). These were distributed among the twelve restoration plantings, six woodland remnants, and three reference sites. We mapped at least one superb fairywren home range in each site, and at least one willie wagtail home range in a site that corresponded to each treatment type. On average, we obtained approximately equal numbers of waypoints for each tracked bird (Table 4.5).

Table 4.5 Mean (± SE) home range (100% minimum convex polygon) of birds tracked in restoration plantings, woodland remnants and large reference sites in the South-west Slopes bioregion of NSW.

Species	Patch type	n	Home range (ha)	Min – max (ha)	No. waypoints
Malurus cyaneus	planting	30	0.30 ± 0.04	0.08 – 0.92	92 ± 10
	remnant	11	0.48 ± 0.16	0.08 – 1.57	110 ± 17
	reference	4	0.60 ± 0.18	0.27 – 1.05	105 ± 41
	all	45	0.37 ± 0.05	0.08 – 1.57	97 ± 8
Rhipidura leucophrys	planting	15	2.32 ± 0.54	0.48 – 8.62	91 ± 14
	remnant	11	1.37 ± 0.24	0.16 – 2.81	94 ± 14
	reference	4	3.09 ± 0.92	1.30 – 5.63	92 ± 15
	all	30	2.07 ± 0.32	0.16 – 8.62	92 ± 9

Home ranges

There was relatively little variation in home range size among the 45 fairywrens we tracked (Table 4.5). In comparison, home ranges of the willie wagtail were highly variable across all site types (Table 4.5).

Home ranges of superb fairywrens were significantly larger in reference sites than in restoration plantings and similarly sized woodland remnants ($p = 0.0343$) (Table 4.6). When assessing the effects of patch size and shape in the latter two site types (i.e. excluding large reference sites), we found that home range size was positively correlated with patch size ($p = 0.00478$) (Table 4.6). The same was true in restoration plantings alone ($p = 0.00296$) (Table 4.6). We did not find any evidence that home range size in the willie wagtail was influenced by patch type, size, or shape. We found no effect of planting age on home range size in either the superb fairywren or the willie wagtail.

The shape of superb fairywren home ranges was strongly dictated by patch shape in restoration plantings and similarly sized remnants ($p < 0.001$) (Table 4.7). Home ranges became more linear as the linearity of patches increased (Figure 4.2). This trend was magnified in restoration plantings ($p < 0.0001$) (Table 4.7; Figure 4.3). We found no effect of patch type or planting age on shape of superb fairywren home ranges. The shape of willie wagtail home ranges was not influenced by patch attributes.

Table 4.6 Parameter estimates for home range size of superb fairywrens, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for all site types (restoration planting, remnant, reference), restoration plantings and remnants, and restoration plantings only. All models that differed from the top model (ΔAIC_c) by ≤2 are shown.

Home range size (all site types)	Rank 1 (w = 0.58)		
	Estimate (SE)		
Intercept	– 1.44 (0.12)		
TYPE (reference)	0.79 (0.34)		
TYPE (remnant)	– 0.10 (0.26)		
Home range size (plantings and remnants)	Rank 1 (w = 0.37)	Rank 2 (w = 0.29)	Rank 3 (w = 0.20)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	– 2.02 (0.21)	– 2.02 (0.20)	– 1.87 (0.25)
SIZE	0.17 (0.05)	0.19 (0.06)	0.16 (0.06)
TYPE (remnant)		– 0.33 (0.24)	
SHAPE			– 0.003 (0.002)
Home range size (plantings only)	Rank 1 (w = 0.39)	Rank 2 (w = 0.26)	Rank 3 (w = 0.20)
	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	– 2.12 (0.20)	– 2.06 (0.20)	– 1.98 (0.24)
SIZE	0.22 (0.06)	0.21 (0.06)	0.21 (0.06)
AGE		0.14 (0.11)	
SHAPE			– 0.003 (0.003)

Table 4.7 Parameter estimates for home range shape of superb fairywrens, ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). Top-ranked models ($\Delta AIC_c \leq 2$) are shown for restoration plantings and remnants, and restoration plantings only. All models that differed from the top model (ΔAIC_c) by ≤2 are shown.

Home range shape (plantings and remnants)	Rank 1 (w = 0.50)	Rank 2 (w = 0.27)
	Estimate (SE)	Estimate (SE)
Intercept	3.69 (0.62)	3.39 (0.62)
SHAPE	0.05 (0.01)	0.04 (0.01)
TYPE (remnant)	– 1.96 (1.03)	
Home range shape (plantings only)	Rank 1 (w = 0.60)	
	Estimate (SE)	
Intercept	3.24 (0.67)	
SHAPE	0.06 (0.01)	

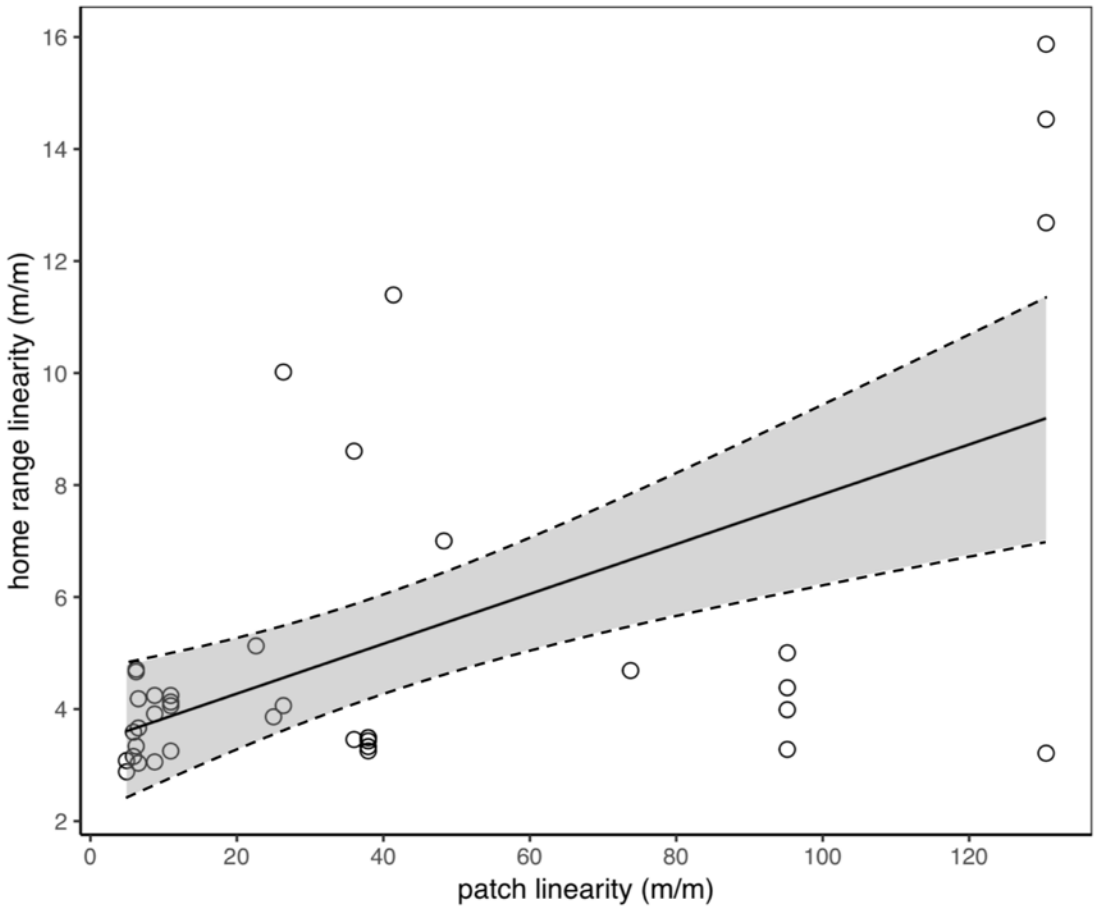

Figure 4.2 The influence of patch shape on home range shape of adult male superb fairywrens (*Malurus cyaneus*) in restoration plantings and similarly sized woodland remnants. Shading indicates 95% confidence intervals. Plot constructed using ggplot2 for R (Wickham 2016).

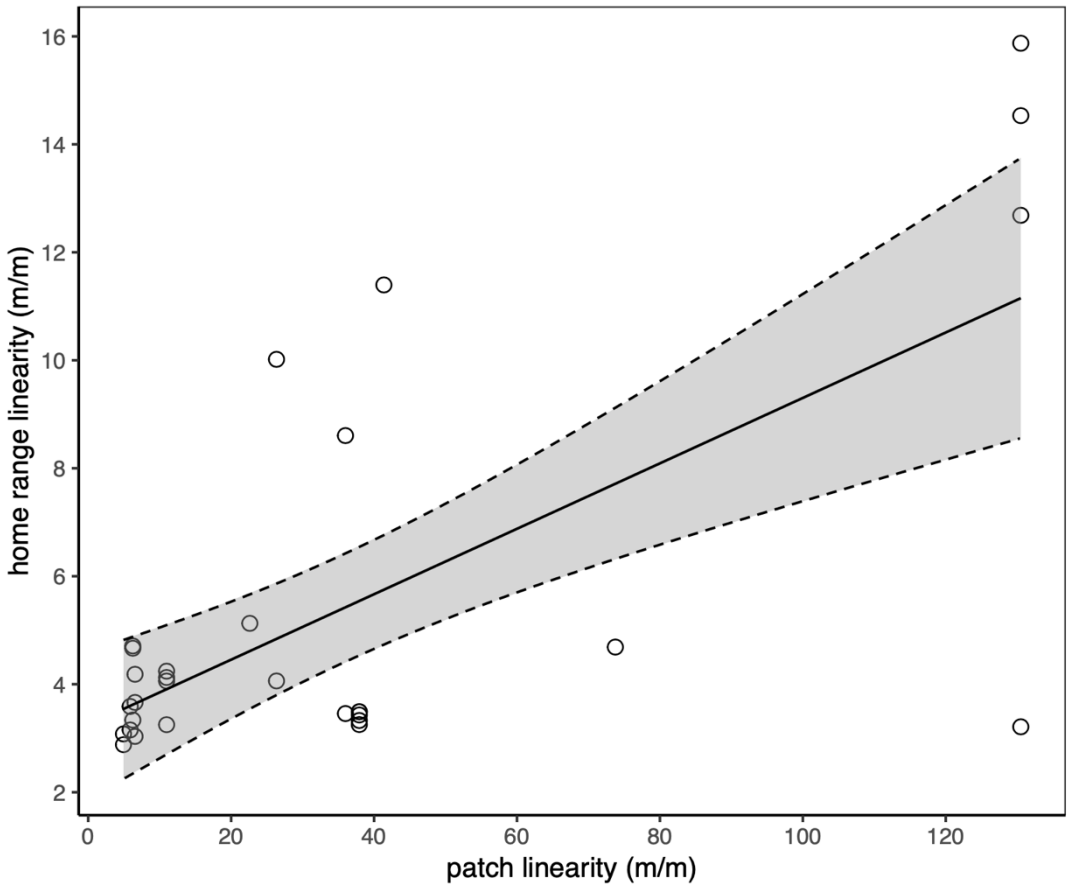

Figure 4.3 The influence of patch shape on home range shape of adult male superb fairywrens (*Malurus cyaneus*) in restoration plantings. Shading indicates 95% confidence intervals. Plot constructed using ggplot2 for R (Wickham 2016).

Movement distances

On average, superb fairywrens moved 13 m between substrates (n=4779, SE=0.21), and willie wagtails moved 26 m (n=3154, SE=0.59). We found no relationship between patch attributes and average movement distances of superb fairywrens or willie wagtails. However, we found that as patch linearity increased in plantings and similarly sized remnants, superb fairywrens were more likely to travel distances of 50 m or more in a single movement, and willie wagtails were more likely to travel 100 m or more in a single movement (Appendix 4.16). We also found that willie wagtails were more likely to move 100 m or more in plantings than in remnants or reference sites (Appendix 4.16).

Gap-crossing

We observed significant gap-crossing behaviour in the willie wagtail. As predicted, individuals in linear sites crossed gaps more frequently than individuals in block-shaped sites. The maximum distance crossed by a single individual was 400 m. This was between two linear restoration plantings on either side of a pasture paddock (Appendix 4.11). The individual responsible for this particular observation crossed this gap twice in a two hour period, using an isolated paddock tree as a stepping stone on the return journey. Another individual crossed a ~100 m gap between two plantings thirteen times during a roughly 3.5-hour tracking period (Appendix 4.12). Superb fairywrens were not observed to cross habitat gaps of more than 35 m while foraging, and rarely crossed gaps at all during observation. However, over the course of the study, we recorded several instances (six confirmed) of superb fairywrens dispersing from one study site to another (see examples in Appendix 4.13–4.15). These individuals would have crossed gaps of up to several hundred metres on the journey between sites.

Discussion

We calculated annual survival for woodland birds in a fragmented agricultural landscape, and documented new information on woodland bird home ranges and movement patterns in restoration plantings and woodland remnants. We discuss our key findings in the remainder of this paper and conclude with some management implications.

Annual survival estimates for the woodland assemblage (51%) and colour-banded birds (55%) were lower than expected. Other Australian studies have commonly reported survival estimates of above 60% for various woodland species in both intact landscapes (Yom-Tov *et al.* 1992; Bridges 1994; Dunn and Cockburn 1999; Green and Cockburn 1999; Gardner *et al.*

2003; Robinson 2008) and fragmented agricultural landscapes (Noske 1991; Zanette 2001; Brooker and Brooker 2001). Survival estimates for the superb fairywren (55%) also were lower than those reported in some previous studies (Yom-Tov *et al.* 1992; Dunn and Cockburn 1999), but within the range reported by Baker *et al.* (1997). Overall, the ostensibly low rate of annual survival in our study sites is concerning, and raises questions about the extent to which restoration plantings are currently contributing to the arresting (and eventual reversal) of woodland bird population declines.

It is possible that restoration plantings act as secondary or transitional habitat for woodland birds, and that retention (and thus apparent survival) of individuals in small patches of revegetation may be poor. However, previous research has shown that various species breed in plantings (Belder *et al.* 2019; 2020), indicating that populations are resident to at least some degree. Annual survival may otherwise be influenced by a range of possible factors, which we were unable to isolate during our study. One potential factor driving low annual survival is the presence of high numbers of predators, especially the introduced red fox (*Vulpes vulpes*), in fragmented agricultural landscapes (Ford *et al.* 2001, Saunders *et al.* 2010, Belder *et al.* 2020).

Unreliable estimates of site fidelity were likely due the low number of capture occasions in our study (J. Laake, pers. comm.). However, we note the high return rate for the rufous whistler, with several colour-banded males returning in consecutive years to defend territories at their initial capture sites. High inter-annual site fidelity has been previously observed in this species in temperate woodlands (Bridges 1994). Our findings are evidence that restoration plantings provide high quality breeding habitat for rufous whistlers.

Question 1. Do patch attributes (type, size, shape) influence survival and site fidelity of woodland birds in fragmented agricultural landscapes?

We did not find strong evidence that site attributes influenced the survival of woodland birds. We postulate that survival may instead be linked to factors that we did not consider in our study, such as predation risk, landscape-scale vegetation cover, connectivity, or climatic variables (Major and Gowing 2001; Whittingham and Evans 2004; Radford *et al.* 2005; Robinson *et al.* 2007, Shanahan *et al.* 2011).

Question 2. How do patch attributes influence the movement patterns of birds, including the size and shape of home ranges?

The average home range size for superb fairywrens (0.37 ha) was smaller than previously reported for the species in fragmented rural habitat (Tidemann 1983; Parsons 2009), and comparable with that reported by Mulder (1992) in high-quality woodland habitat. Previous research has identified that this species is a "planting specialist", and will preferentially occupy restoration plantings over other kinds of habitat in fragmented agricultural landscapes (Belder *et al.* 2018). It is possible that the home range sizes we observed in our study indicate that restoration plantings provide good quality habitat for the superb fairywren. However, a small home range in a fragmented landscape may also be indicative of reduced foraging efficiency associated with low habitat availability (Hinam and Clair 2008). This may also explain why the home range size of the superb fairywren was positively associated with patch size – larger sites permit the expansion of home ranges. Previous research has documented more bird breeding activity in smaller plantings and woodland remnants (Belder *et al.* 2019). Superb fairywren home ranges in smaller patches may also be constrained by higher densities of breeding individuals.

We could find only one published study reporting on the average home range size of willie wagtails – breeding pairs in Papua New Guinea occupied average home ranges of 0.85 ha (Dyrcz 1994). This is substantially smaller than the average home range size of willie wagtails in our study (2.07 ha), and may be linked to the intactness of the landscape in which the study took place. We suggest that willie wagtails in our study maintained larger home ranges than they would in intact landscapes. The significant gap-crossing behaviour that we observed (discussed further in subsequent paragraphs) is further evidence of this.

Superb fairywrens that resided in linear sites had much more elongate home ranges than those in block-shaped sites. According to optimal foraging theory, an elongate home range does not facilitate optimal foraging – as the maximum distance from a central point increases, so too does the energy required to reach that point (Andersson 1978; Pyke 2010). It is also more difficult for an individual to patrol and defend an elongate territory, as the outer extremities are further away from the core of the territory (Dill 1978). Both superb fairywrens and willie wagtails were more likely to travel long distances between substrates while foraging in linear sites – evidence that energy expenditure is indeed higher in these sites than in block-shaped sites. We suggest that linear sites constrain the home range shape of woodland birds, potentially placing individuals in these patches at a disadvantage compared to individuals in block-shaped sites. This is particularly relevant for breeding birds, although previous research has found no effect of patch linearity on breeding success (Belder *et al.* 2019 unpublished work).

Previous studies have found that linear sites support lower overall species richness and abundance than similarly sized block-shaped sites (Lindenmayer *et al.* 2010b). Our results add weight to the idea that block-shaped sites better support woodland birds than linear sites.

However, we acknowledge the importance of linear sites in improving landscape connectivity in fragmented agricultural landscapes (Schippers *et al.* 2009; Lentini *et al.* 2011). Furthermore, intersecting linear patches have been found to support similar species richness and abundance to block-shaped sites (Lindenmayer *et al.* 2007). These findings highlight the need to consider the context and geographical location of restoration plantings in conservation planning. For instance, it may be beneficial to establish new plantings near to existing vegetation, widen linear strips, fill in corners where strips meet, or link small remnants.

The gap-crossing behaviour that we documented in the willie wagtail is also of interest in the context of restoration planting configuration in fragmented agricultural landscapes. For individuals that frequently cross large habitat gaps, a high proportion of the territory is matrix habitat (pasture or cropland), and unsuitable for foraging. These individuals therefore expend considerably more energy commuting between areas of high quality habitat within their home range than do individuals who maintain home ranges entirely or mostly within a single habitat patch. Previous research has identified that woodland birds needing to cross large habitat gaps experience lower reproductive success and output (Hinsley *et al.* 2008). The configuration and isolation of many restoration plantings may increase the energy expenditure of breeding birds, and therefore add to the existing pressures that breeding birds face in these landscapes. Furthermore, species that are unwilling or unable to cross large habitat gaps are likely to be disadvantaged by a lack of landscape connectivity (Brooker *et al.* 1999; Lees and Peres 2009; Garrard *et al.* 2012). Poor landscape connectivity has been linked to local extinctions of woodland and forest bird species (Bellamy *et al.* 1996; Uezu *et al.* 2005; Doerr *et al.* 2011; Pavlacky *et al.* 2012).

Willie wagtails visited scattered paddock trees within the agricultural matrix, particularly while crossing habitat gaps. Such behaviour has been observed in other small to medium-sized woodland birds in fragmented landscapes (Fischer and Lindenmayer 2002a; Robertson and Radford 2009). Paddock trees (and other isolated habitat features) facilitate connectivity in fragmented agricultural landscapes (Manning *et al.* 2006). These features are of disproportionate importance in ensuring the persistence of fauna populations in these landscapes (Gibbons and Boak 2002; Le Roux *et al.* 2015; Le Roux *et al.* 2018).

Inferential limitations and future research directions

This study provides novel insights into survival and home ranges of woodland birds in fragmented agricultural landscapes. However, we note the short duration of the study and low number of physical recapture efforts per site, acknowledge that long-term trends cannot be forecast from a study of only three years. Furthermore, we are unable to infer the causes of an apparently high rate of annual turnover – it is unclear whether this was caused by emigration, mortality, or a combination of the two processes. The presence of site attributes in top-ranked survival models indicates that these variables may influence survival to some degree. Alternatively, factors outside the scope of this study may be more influential. Future research should focus on investigating this.

We encourage researchers to undertake long-term bird banding studies in fragmented agricultural landscapes. While banding requires considerable resources and effort, it provides invaluable information on the survival, movement, persistence, and activities of individual birds and their populations. A national network of banding projects would improve understanding of bird movements and demographic processes. Advancements in telemetry techniques may facilitate the collection of these data in the future.

We also suggest conducting home range and movement studies on a wider range of woodland bird species, especially those which may be more dependent on tree cover.

Concluding remarks

The value of small habitat patches for biodiversity conservation has been well-documented and reviewed (Fischer and Lindenmayer 2002b; Tulloch *et al.* 2016; Lindenmayer 2019; Wintle *et al.* 2019). Restoration plantings provide habitat that is complementary to existing woodland remnants (Ikin *et al.* 2018), and in which birds can breed successfully (Belder *et al.* 2020). There is no doubt that restoration plantings are a necessary conservation strategy in fragmented agricultural landscapes. However, the results of our study indicate that 1) patch isolation and certain patch configurations place resident and breeding birds at an energetic disadvantage, and 2) in our study area, woodland bird populations are continuing to decline, and poor annual survival may contribute to this. Urgent action is needed to restore habitat at a landscape scale if we are to avoid ongoing losses and local extinctions of woodland birds in southern Australia.

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Appendix 4.1 Numbers of woodland birds banded in study sites in the Southwest Slopes Bioregion, NSW. Numbers are given as "total banded (no. recaptures)". Species that were colour-banded are underlined. Species are listed in taxonomic order (Gill and Donsker 2018). Note that the white-plumed honeyeater was frequently captured in all sites, but only banded on one occasion in one site.

Common name	Species	Planting			Remnant			Reference			Total	Retrap
		2015	2016	2017	2015	2016	2017	2015	2016	2017		
white-throated treecreeper	*Cormobates leucophaea*					1	1	1		2	5	
brown treecreeper[CV]	*Climacteris picumnus*				3	6	9 (2)	2	2	1	23	2
superb fairywren	*Malurus cyaneus*	148	158 (51)	98 (17)	42	39 (12)	22 (6)	14	8 (2)	11 (2)	540	90
black-chinned honeyeater[CV]	*Melithreptus gularis*					2					2	
white-plumed honeyeater	*Ptilotula penicillata*				7		1 (1)				8	1
spotted pardalote	*Pardalotus punctatus*	3							1		4	
striated pardalote	*Pardalotus striatus*	1	1			1					3	
speckled warbler[C]	*Pyrrholaemus sagittatus*		1	5	2		1	3		2	14	
white-browed scrubwren	*Sericornis frontalis*	2	1	5 (1)							8	1
weebill[C]	*Smicrornis brevirostris*	2	9 (1)	3 (1)	2		1 (1)	5	1		23	3
western gerygone	*Gerygone fusca*	1			2						3	
brown thornbill	*Acanthiza pusilla*	4	4 (2)	1 (1)				2		1 (1)	12	4
buff-rumped thornbill	*Acanthiza reguloides*	5	2	2	8	6 (1)	6 (1)	8	5 (1)	14 (4)	56	7
yellow-rumped thornbill[C]	*Acanthiza chrysorrhoa*	52	65 (7)	31 (12)	5	14 (1)	4	2	11	3	187	20
yellow thornbill	*Acanthiza nana*	29	34 (5)	15 (2)	3	4	2 (1)	1			88	8
striated thornbill	*Acanthiza lineata*	3	3 (1)	3 (1)		2	6		15	2 (2)	34	4
white-browed babbler	*Pomatostomus superciliosus*	12	4 (2)	9 (3)		9	6 (1)				40	6
dusky woodswallow[CV]	*Artamus cyanopterus*	1									1	
white-winged triller[C]	*Lalage tricolor*	2									2	
crested shriketit[C]	*Falcunculus frontatus*		1		1	1	2				5	
golden whistler	*Pachycephala pectoralis*	4	1	5	3	4	1				18	
rufous whistler	*Pachycephala rufiventris*	8		1	7						16	
grey shrikethrush	*Colluricincla harmonica*	9	14	12	2	5	2			1	45	
willie wagtail	*Rhipidura leucophrys*	8	7	5 (1)	7	5	6 (2)	1			39	3
grey fantail	*Rhipidura albiscapa*	14	1	5		1	2	2			25	

Common name	Species	Planting			Remnant			Reference			Total	Retrap
		2015	2016	2017	2015	2016	2017	2015	2016	2017		
restless flycatcher[C]	*Myiagra inquieta*					2					2	
eastern yellow robin	*Eopsaltria australis*	1		1				1		2	5	
jacky winter[C]	*Microeca fascinans*				1			2	1 (1)		4	1
flame robin[CV]	*Petroica phoenicea*	6	28	15		8	1				58	
scarlet robin[CV]	*Petroica boodang*	2	4	3		4	2 (1)	1	1	3	20	1
red-capped robin[C]	*Petroica goodenovii*	2	5 (1)	2			1	3			13	1
rufous songlark	*Cincloramphus mathewsi*	3			3						6	
silvereye	*Zosterops lateralis*								2		2	
common blackbird[I]	*Turdus merula*		4	1		1	1				7	
diamond firetail[CV]	*Stagonopleura guttata*	3	5	2			1				11	
red-browed finch	*Neochmia temporalis*	5	18	30 (1)	3	12	10				78	1
double-barred finch	*Taeniopygia bichenovii*		2								2	
European goldfinch[I]	*Carduelis carduelis*			1							1	

Appendix 4.2 Numbers of woodland birds colour-banded in 2015 and 2016 in the South-west Slopes bioregion.

Species	Common name	Colour-banded 2015	Colour-banded 2016	Total colour-banded
Malurus cyaneus	superb fairywren	204	140	344
Acanthiza chrysorrhoa	yellow-rumped thornbill	63	82	145
Acanthiza reguloides	buff-rumped thornbill	21	11	32
Colluricincla harmonica	grey shrikethrush	11	19	30
Pachycephala rufiventris	rufous whistler	15	-	15
Pomatostomus superciliosus	white-browed babbler	12	3	15
Rhipidura leucophrys	willie wagtail	-	12	12
Petroica goodenovii	red-capped robin	5	4	9
Stagonopleura guttata	diamond firetail	3	5	8
Pyrrholaemus sagittatus	speckled warbler	5	1	6

Appendix 4.3 Top model for annual survival of the woodland assemblage in restoration plantings, similarly sized woodland remnants and large reference sites, using the CJS model.

Model description	npar	$QAIC_c$	ΔAIC_c	AIC_w	Deviance
ϕ(.) p(.)	3	82.67	0.00	0.77	29.10

Appendix 4.4 Top models for annual survival of the woodland assemblage in restoration plantings and similarly sized woodland remnants (excluding large reference sites), using the CJS model.

Model description	npar	$QAIC_c$	ΔAIC_c	AIC_w	Deviance
ϕ(.) p(.)	3	76.76	0.00	0.22	33.47
ϕ(.) p(type)	4	78.69	1.93	0.09	866.16
ϕ(.) p(shape)	4	78.75	1.99	0.08	866.91

Appendix 4.5 Top models for annual survival of colour-banded birds in restoration plantings, similarly sized woodland remnants and large reference sites, using the CJS and Barker models.

Model description	npar	$QAIC_c$	ΔAIC_c	AIC_w	Deviance
CJS model					
ϕ(.) p(.)	3	16.04	0.00	0.78	28.42
Barker model					
S(.) p(.) R(.) F(.) F'(.)	5	45.11	0.00	0.68	1772.31

Appendix 4.6 Top models for annual survival of colour-banded birds in restoration plantings and similarly sized woodland remnants (excluding large reference sites), using the CJS and Barker models.

Model description	npar	$QAIC_c$	ΔAIC_c	AIC_w	Deviance
CJS model					
ϕ(.) p(.)	3	71.95	0.00	0.23	33.29
ϕ(size) p(.)	4	73.87	1.92	0.09	664.99
Barker model					
S(.) p(.) R(.) F(.) F'(.)	5	152.48	0.00	0.15	1744.83
S(type) p(.) R(.) F(.) F'(.)	6	154.42	1.94	0.06	3447.08
S(.) p(type) R(.) F(.) F'(.)	6	154.44	1.95	0.06	3447.39
S(.) p(.) R(.) F(size) F'(.)	6	154.46	1.98	0.05	3447.91
S(shape) p(.) R(.) F(.) F'(.)	6	154.47	1.99	0.05	3448.27

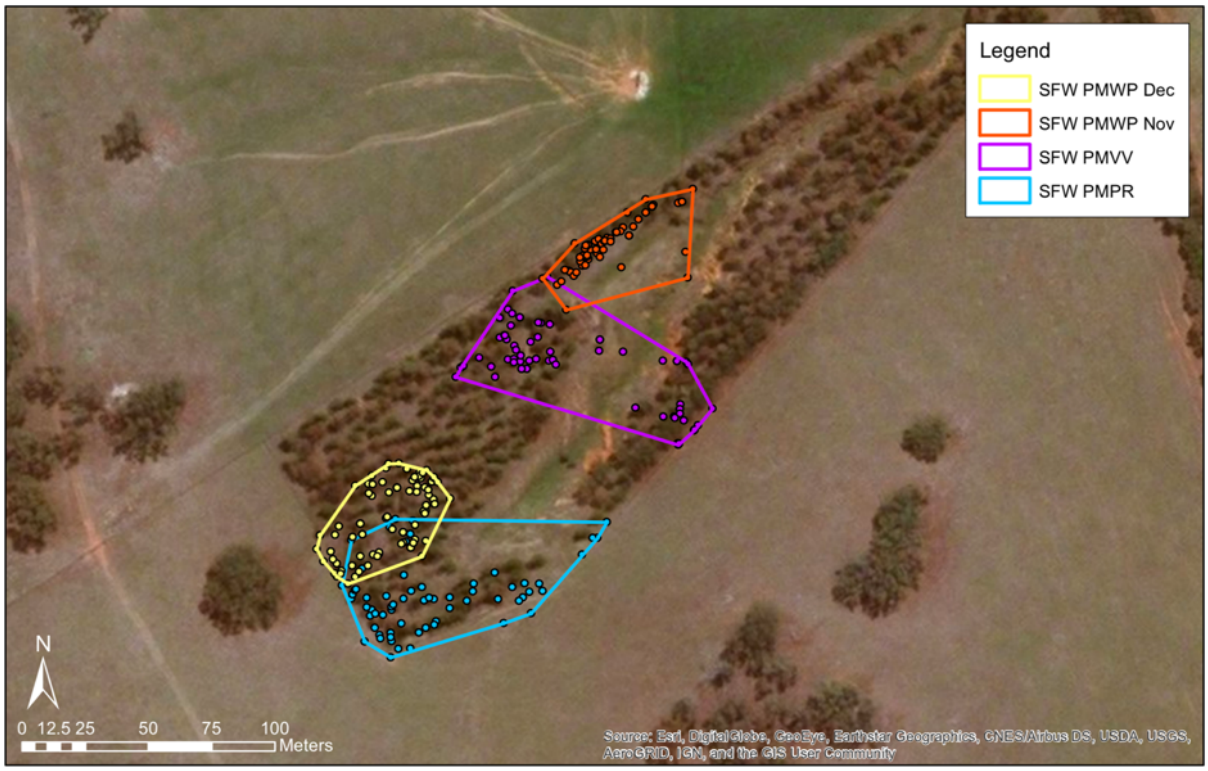
Appendix 4.7 Top models for annual survival of the superb fairywren (*Malurus cyaneus*) in restoration plantings, similarly sized woodland remnants and large reference sites, using the CJS and Barker models.

Model description	npar	$QAIC_c$	ΔAIC_c	AIC_w	Deviance
CJS model					
ϕ(.) p(.)	3	16.12	0.00	0.76	35.37
Barker model					
S(.) p(.) R(.) F(.) F'(.)	5	41.21	0.00	0.63	399.36

Appendix 4.8 Top models for annual survival of the superb fairywren (*Malurus cyaneus*) in restoration plantings and similarly sized woodland remnants (excluding large reference sites), using the CJS and Barker models.

Model description	npar	$QAIC_c$	ΔAIC_c	AIC_w	Deviance
CJS model					
ϕ(.) p(.)	3	71.95	0.00	0.23	33.29
ϕ(size) p(.)	4	73.87	1.92	0.09	664.99
Barker model					
S(.) p(.) R(.) F(.) F'(.)	5	152.48	0.00	0.15	1744.83
S(type) p(.) R(.) F(.) F'(.)	6	154.42	1.94	0.06	3447.08
S(.) p(type) R(.) F(.) F'(.)	6	154.44	1.95	0.06	3447.39
S(.) p(.) R(.) F(size) F'(.)	6	154.46	1.98	0.05	3447.91
S(shape) p(.) R(.) F(.) F'(.)	6	154.47	1.99	0.05	3448.27

Appendix 4.9 Home ranges of male superb fairywrens (*Malurus cyaneus*) in a linear restoration planting in the South-west Slopes bioregion, NSW.



Appendix 4.10 Home ranges of male superb fairywrens (*Malurus cyaneus*) in a block-shaped restoration planting in the South-west Slopes bioregion, NSW.

Appendix 4.11 Movements of a willie wagtail (01FA) within and between adjacent restoration plantings in the South-west Slopes bioregion.

Appendix 4.12 Movements of a willie wagtail (11S5) within and between adjacent restoration plantings in the South-west Slopes bioregion.

Appendix 4.13 Spatial arrangement of sites in the Morven area, NSW, in which inter-patch movement (dispersal) of colour-banded superb fairywrens was recorded during a three-year mark-recapture-resight study. Arrows indicate the direction(s) of movement.

Appendix 4.14 Spatial arrangement of sites on the property “Burnbank”, NSW, in which inter-patch movement (dispersal) of colour-banded superb fairywrens was recorded during a three-year mark-recapture-resight study. The arrow indicates the direction of movement.

Appendix 4.15 Spatial arrangement of sites on the property “Wisconsin”, NSW, in which inter-patch movement (dispersal) of colour-banded superb fairywrens was recorded during a three-year mark-recapture-resight study. The arrow indicates the direction of movement.

Appendix 4.16 Parameter estimates for longer movement distances of superb fairywrens (>50 m) and willie wagtails (>100 m), ranked by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). All models that differed from the top model (ΔAIC_c) by ≤2 are shown.

SFW: movement distance >50 m	Rank 1 (*w* = 0.14)	Rank 2 (*w* = 0.13)	Rank 3 (*w* = 0.10)	Rank 4 (*w* = 0.08)	Rank 5 (*w* = 0.08)
	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)	Estimate (SE)
Intercept	0.01 (0.0009)	0.02 (0.0005)	0.01 (0.001)	0.02 (0.0005)	0.01 (0.0005)
SHAPE	– 0.0003 (0.000009)	– 0.0003 (0.000009)	– 0.0003 (0.000009)	– 0.00007 (0.00004)	– 0.00002 (0.000009)
SIZE	0.0003 (0.0002)		0.0002 (0.0002)		
TYPE (remnant)	– 0.001 (0.0009)	– 0.001 (0.0009)	– 0.003 (0.002)	– 0.001 (0.0009)	
SHAPE:TYPE (remnant)	0.00005 (0.00002)	0.00005 (0.00002)	0.00005 (0.0002)	0.00004 (0.00002)	
SIZE:TYPE			0.0006 (0.0005)		
SHAPE:SIZE				0.00001 (0.00001)	

SFW: movement distance >50 m (cont.)	Rank 6 (*w* = 0.07)	Rank 7 (*w* = 0.07)
	Estimate (SE)	Estimate (SE)
Intercept	0.01 (0.0009)	0.02 (0.0005)
SHAPE	– 0.00002 (0.000009)	– 0.00008 (0.00004)
SIZE	0.0003 (0.0002)	
SHAPE:SIZE		0.00002 (0.00001)

WW: movement distance >100 m	Rank 1 (*w* = 0.28)	Rank 2 (*w* = 0.12)
	Estimate (SE)	Estimate (SE)
Intercept	0.008 (0.0003)	0.008 (0.0004)
SHAPE	– 0.00002 (0.0000005)	– 0.00002 (0.0000005)
SIZE		– 0.00007 (0.0001)
TYPE (remnant)	0.0002 (0.0006)	
SHAPE:TYPE	0.00002 (0.00001)	0.00003 (0.000007)

WW: movement distance >100 m (all sites)	Rank 1 (*w* = 0.75)
	Estimate (SE)
Intercept	0.007 (0.0002)
TYPE (remnant)	0.001 (0.0004)
TYPE (reference)	0.00009 (0.0006)

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CONCLUSIONS

The establishment of restoration plantings is an increasingly common conservation strategy in fragmented agricultural landscapes worldwide (Rey Benayas *et al.* 2009; Barral *et al.* 2015; Crouzeilles *et al.* 2016). The ecological benefits of restoration plantings are numerous, but are often poorly quantified, particularly in the long-term (Block *et al.* 2001; Hobbs 2003; Ruiz-Jaen and Aide 2005). This limits the ability of land managers and conservation scientists to assess whether restoration plantings are succeeding as a conservation strategy, particularly for fauna of conservation concern (Ruiz-Jaen and Aide 2005; Belder *et al.* 2018).

My thesis aimed to address a significant knowledge gap associated with the population responses of woodland birds – a species assemblage of conservation concern – to restoration plantings in a fragmented agricultural landscape. By conducting a detailed, population-oriented study, using a range of techniques that moved beyond commonly used metrics such as presence and abundance, I have provided new insights into the suitability of restoration plantings as habitat for declining woodland birds. I documented breeding activity and quantified breeding success, estimated annual survival via a mark-recapture-resight study, and examined individual birds' home range sizes and movement patterns. The series of papers presented in this thesis represent an advancement of our understanding of woodland bird responses to restoration plantings in fragmented agricultural landscapes. My findings have direct management implications, and the knowledge gained from my research can be readily applied to conservation planning.

Prevalence of pattern-based data in Australia

In chapter 1, I synthesised current knowledge of birds in restoration plantings and fragmented agricultural landscapes. I identified that the majority of previous research has focused on

pattern-based metrics such as bird species richness and abundance to infer habitat quality in restoration plantings. A general observation is that population-oriented research has been conducted much more extensively in regions such as North America and Europe than in Australia. In light of historic and ongoing declines of wildlife (Woinarski *et al.* 2015; Geyle *et al.* 2018), Australian land managers have much to gain from an increase in mechanistic studies on fauna of conservation concern. Research questions that I propose as priorities for future research on birds in restoration plantings centre on the provision of food resources, formation of optimal foraging patterns, nest-predation levels and the prevalence of primary predators, the role of brood parasitism, and the effects of patch size and isolation on resource availability and population dynamics in a restoration context.

Value of small patches for breeding birds

A key aim of my study was to identify whether woodland birds can breed successfully in restoration plantings. The short answer is yes. However, a number of the findings presented in chapters 2 and 3 should be considered. First, I found that bird species' relative abundance was not predictive of their degree of breeding activity. This finding highlights the risk of continuing to rely on simple metrics such as abundance to draw conclusions about habitat quality, and more importantly, the apparent success of conservation strategies.

Second, I found that smaller patches supported both more breeding activity, and more successful breeding by woodland birds, than larger patches. Although I was unable to identify the underlying cause(s) of this trend, it is nonetheless of interest for conservation planning. My findings are a novel contribution to the substantial body of evidence emphasising the value of small habitat patches for wildlife conservation (Manning *et al.* 2006; Le Roux *et al.* 2015; Tulloch *et al.* 2016; Lindenmayer 2019; Wintle *et al.* 2019).

Influence of nest-predators on conservation outcomes

Through nest-monitoring, I identified two major types of nest predator in my study sites: native predatory birds (corvids and artamids), and the introduced European red fox. Bird species such as the Australian magpie, pied and grey butcherbirds, pied currawong, and Australian and little ravens, are natural predators of Australian bird nests that have benefited from habitat fragmentation (Madden *et al.* 2015; Fulton 2018). A review by Smith *et al.* (2010) suggested that predator control is an effective conservation strategy for bird populations of conservation concern. It may be possible to control for avian predators via targeted culls (e.g. Debus 2006). However, culling overabundant bird species is often difficult and rarely produces long-term benefits (Guillemette and Brousseau 2001; Rothstein and Peer 2005; O'Loughlin *et al.* 2017; Beggs *et al.* 2019).

The European red fox is an exotic predator that is a major contributor to the decline of wildlife in southern Australia (Kinnear *et al.* 2002; Mahon 2009; Braysher 2017). Previous research has focused on the effects of foxes on critical weight range mammals (see Woinarski *et al.* 2015). My research indicates that foxes also may be contributing to woodland bird population declines.

The prevalence of foxes in Australian agricultural landscapes means that birds nesting in both restoration plantings and woodland remnants are vulnerable to fox predation. Foxes may therefore be reducing the efficacy of restoration plantings in conserving and restoring woodland bird populations. Fox control is widely implemented across southern Australia (Mahon 2009). However, the intensity of control efforts are not usually sufficient to have a meaningful impact on fox abundance (Gentle *et al.* 2007). When concerted and prolonged fox control programs are in place, it is possible to achieve positive outcomes for at-risk fauna

(Priddel and Wheeler 1997; Dexter *et al.* 2007; Dexter and Murray 2009; Carter 2010). Conservation fencing, which protects against both foxes and feral cats, could also be considered for larger restoration projects (De Tores and Marlow 2012; Dickman 2012; Ringma *et al.* 2017).

My research did not uncover any evidence of nest-predation by feral cats. A cautiously optimistic interpretation is that feral cats pose less of a threat to nesting woodland birds in my study region than do foxes. However, feral cats are known to rapidly switch between prey types/species depending on resource availability (Doherty *et al.* 2015), and individual feral cats may also specialise in particular prey species (Dickman and Newsome 2015). As the impacts of feral cats vary in time and space, and according to the individual cats present in an area at a given time, land managers should not assume that feral cat predation is non-existent in the South-west Slopes, or in fragmented agricultural landscapes more generally.

Influence of patch geometry on bird movements

The home ranges of a small, sedentary passerine – the superb fairywren – were closely linked to patch size and shape in my study. In particular, home ranges in linear sites were constrained by patch edges, resulting in increasingly linear home ranges as patch linearity increased. Since the extremities are further apart, linear home ranges do not promote optimal foraging, and are more energetically expensive to maintain (Stephens 2008). Although I found no influence of site shape on survival or breeding success, it is worth considering that woodland birds that are resident in linear patches may be at an energetic disadvantage compared with those resident in block-shaped sites. Linear patches are important for landscape connectivity (Schippers *et al.* 2009; Lentini *et al.* 2011), but I recommend prioritising the implementation of block-shaped sites as much as possible. Improving

connectivity between adjacent patches may also reduce the need for extensive gap-crossing, such as that observed in the willie wagtail.

Ongoing declines of woodland birds

My mark-recapture-resight study in restoration plantings and remnant woodland patches was the first of its kind in an Australian fragmented agricultural landscape. Annual survival estimates from the study revealed an ostensibly high degree of turnover due to mortality and/or emigration. Traditional presence/abundance surveys may not detect low survival rates if the number of individuals recorded in a given site is approximately equal over time. However, many Australian birds are long-lived, meaning that the demographic effects of poor survival and recruitment may not be observed until older individuals begin to die. This phenomenon has been observed in vegetation communities in Australia's arid lands as a result of grazing by introduced herbivores (Auld *et al.* 2015). Given this knowledge, I suggest that there is substantial risk in not closely monitoring survival and persistence in woodland bird populations. Further, adaptive management is facilitated by effective monitoring of fauna of conservation concern (Nichols and Williams 2006; Lindenmayer and Likens 2018). Long-term mark-recapture-resight studies would improve the understanding of the demographic processes of woodland bird species, and may enable accurate estimates of site fidelity (which the short duration of my study did not permit). It would also be useful to further investigate the potential influence of site attributes on survival and site fidelity.

The low annual survival rate that I found in my study should be interpreted with caution, given that only three years of mark-recapture-resight data were used to calculate the estimate. However, my findings are consistent with a known trend towards population decline of woodland bird assemblages (Barrett *et al.* 2007; Lindenmayer and Cunningham 2011; Rayner

et al. 2014; Lindenmayer *et al.* 2018). I suggest that the low annual survival rate in my study sites may be an artefact of a struggling regional population, rather than a reflection of habitat quality in restoration plantings. The historic cause of woodland bird declines is loss of habitat (Ford 2011), and land clearing in south-eastern Australia has not ceased (Bradshaw 2012; Evans 2016). Indeed, environmental legislation in New South Wales has facilitated land clearing in agricultural regions (Bartel and Graham 2016). The amount of habitat that has been restored through revegetation activities is unlikely to have been sufficient to halt or reverse population declines. Landscape-scale habitat restoration is necessary to achieve meaningful conservation outcomes for woodland birds.

Limitations of the research and future recommendations

The findings described in this thesis should be interpreted with the following considerations. First, a study of duration only three years is unlikely to capture a significant degree of inter-annual variation in breeding success, individual mortality, and annual turnover in woodland bird populations. The limitations of short-term studies on bird communities are discussed by Maron *et al.* (2005).

The two years in which I completed major field seasons for this study were both years that recorded above average rainfall in the study region (Bureau of Meteorology 2019). In September and October 2016, the Murrumbidgee River flooded, peaking at 8.3 m (Bureau of Meteorology 2017). Some areas were so wet that by October, reaching the sites required a 3 km return trip on foot. Some sites remained that way until late November. As such, it is possible that the breeding season recorded in 2016 was more productive than may be the case according to long-term trends (Gibbs *et al.* 2011). In light of the recent (2018-2019) drought in NSW, it would be worth repeating aspects of my work to establish a) how bird breeding

success compares in dry years, and b) whether restoration plantings act as drought refugia for woodland birds.

It is important to consider the projected effects of climate change on woodland bird communities. Much of Australia is projected to become increasingly hot and dry (CSIRO and Bureau of Meteorology 2015), which may have a significant negative effect on bird breeding (Gibbs *et al.* 2011). It is therefore important to implement adaptive management of restoration plantings to ensure that they provide optimal habitat for woodland birds in a changing climate.

Further to this, climate and weather may not vary in the same way in other ecosystems. For example, tropical rainforest or heathland may not experience the same maturation trajectories for vegetation restoration as temperate woodlands, nor may they experience the same degree of inter-annual or intra-annual variation in weather and climate. Breeding seasons for bird communities in other ecosystems, such as arid woodlands or shrublands, may be responsive to unpredictable environmental cues such as significant rainfall events. I advise caution in applying the results presented in this thesis to other restored landscapes or ecological communities.

An additional limitation of the study is that, due to the species accumulation curve, the majority of data in the study are attributed to the more common and abundant species in the study region. I recommend that studies be undertaken to examine more individual species. This is because species vary widely in their habitat requirements and their responses to fragmentation and habitat loss. In particular, the responses of woodland-dependent species may differ to those of more common species that occupy a broader range of habitat types.

The woodland patches in which this study was conducted, including the "large reference sites", were smaller in size than those used in other studies to infer the effects of patch size on woodland bird community composition, population dynamics, and ecosystem processes. The South-west Slopes bioregion contains very little remnant woodland habitat, and the references sites chosen for my study were among the largest available. However, it is important that the patches of "intact" woodland selected in this highly degraded landscape are not normalised as a baseline. I recommend that studies be undertaken in landscapes that permit the inclusion of truly intact (>500 ha) woodland remnants.

I strongly recommend that studies such as those presented in this thesis be undertaken over much longer time periods, and incorporated into long-term monitoring programs whenever possible. As discussed previously, mechanistic and population-based studies provide crucial insights into how woodland birds are responding to revegetation and other restoration efforts. This is particularly relevant in light of my finding that the relative abundance of individual bird species is not necessarily correlated with their degree of breeding activity. These differences are of concern to land managers who need to use limited funding to ensure that conservation strategies are achieving conservation outcomes in the long-term. The only effective way to ensure inter-annual variation is captured is to include these studies in long-term monitoring.

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APPENDIX A

Likely depredation of a Superb Fairy-wren *Malurus cyaneus* nest by an uncommon predator, the White-browed Babbler *Pomatostomus superciliosus*

A white-browed babbler depredating a superb fairywren nest. Photo: Donna Belder.

Belder, D. J. (2018). Likely depredation of a Superb Fairy-wren *Malurus cyaneus* nest by an uncommon predator, the White-browed Babbler *Pomatostomus superciliosus*. *Australian Field Ornithology* **35**, 146–148.

Abstract

This paper describes a likely incident of opportunistic depredation of a Superb Fairy-wren *Malurus cyaneus* nest by an uncommon predator, the White-browed Babbler *Pomatostomus superciliosus*, and presents the first photographic evidence of nest-depredation by this species. Potential implications of the observation are discussed in the context of detecting unexpected and uncommon nest-predators.

Introduction

Nest-depredation is a significant pressure for breeding birds and causes up to 90% of failed nesting attempts (Guppy *et al.* 2017; Okada *et al.* 2017). Major nest-predators in Australian temperate woodlands include the introduced Red Fox *Vulpes vulpes*, Eastern Brown Snake *Pseudonaja textilis*, rodents, and various predatory bird species (Zanette and Jenkins 2000; Debus 2006; Colombelli-Négrel and Kleindorfer 2009; D. Belder unpubl. data). Avian nest-predators typically include large, predominantly carnivorous species such as butcherbirds *Cracticus* spp., the Australian Magpie *Gymnorhina tibicen*, currawongs *Strepera* spp., and corvids *Corvus* spp. (Zanette and Jenkins 2000; Debus 2006). However, unexpected and/or opportunistic nest-predators such as the Brown-headed Honeyeater *Melithreptus brevirostris* (Zanette 1997), Eastern Spinebill *Acanthorhynchus tenuirostris* (Guppy *et al.* 2016), and Eastern Whipbird *Psophodes olivaceus* (Guppy *et al.* 2017) have also been reported. This paper adds to a growing body of evidence that a wide range of omnivorous bird species may prey on the eggs of other species and documents an unexpected nest-predator in the context of box-gum grassy woodland habitat restoration.

Study area and methods

I located a Superb Fairy-wren *Malurus cyaneus* nest in a box–gum grassy woodland restoration planting on a farm near Wagga Wagga, New South Wales, on 21 October 2016. The nest was situated 0.35 m above the ground in a patch of Black Anther Flax-lily *Dianella revoluta*, and was under construction at the time of discovery. A UOVision UV565HD motion-sensing wildlife camera was installed ~1.5 m from the nest on the day of discovery. This was set to normal sensitivity, with a burst of three images to be captured at a minimum interval of 30 seconds, and was running constantly from the time of installation. The nest was again checked on 25 October, 31 October, and 4 November. Three eggs were laid between 31 October and 4 November. On 9 November, the nest was found to be empty but intact. I reviewed all camera imagery recorded between 4 and 9 November and identified the date, time, and probable cause of the nest failure.

Results

Imagery captured by the wildlife camera on 8 November 2016 shows a group of White-browed Babblers *Pomatostomus superciliosus* foraging amongst the Black Anther Flax-lily in which the Superb Fairy-wren nest was located. The Babblers are first captured in a sequence of images taken at 0706 h Australian Eastern Daylight Time. In the next sequence of images, at 0707 h, a single Babbler is seen with its head in the nest (Figure A1). Two breeding-plumaged male Fairy-wrens on either side of the nest appear to be exhibiting distressed or defensive behaviour; one shows the flattened posture and fluffed feathers typical of a defensive display. The camera subsequently captured at least three Superb Fairy-wrens (at least two males and one female) returning to the nest until 0840 h, after which time no further activity was recorded at the nest. The nest was known to be between 4 and 6 days into the incubation phase at the time of depredation.

Figure A1 White-browed Babbler apparently depredating a Superb Fairy-wren nest, 8 November 2016. The camera viewed the nest from the right-hand side. Note distressed/defensive behaviour exhibited by two adult male Fairy-wrens. Photo: Donna J. Belder.

Discussion

The recorded imagery depicts a White-browed Babbler probing a Superb Fairy-wren nest. This behaviour could be attributed either to predation on eggs or young, or removal of nesting material. As the nest appeared to be intact when it was subsequently checked, and no egg fragments were observed in the vicinity of the nest, it is doubtful that the Babbler inadvertently dislodged the eggs while stealing nesting material. It is therefore likely that the imagery depicts an incident of opportunistic depredation.

White-browed Babblers are typically classed as insectivores, and primarily forage among leaf-litter on the ground (Antos and Bennett 2006). Their diet consists largely of invertebrates, but they are also known to feed on small amphibians and reptiles, as well as fruits and seeds (Higgins and Peter 2002). This indicates that this species is not strictly insectivorous, and will opportunistically select other high-value prey items. There are two published records of White-browed Babblers depredating bird nests in woodland sites: Carter (1923) documented consumption of Red Wattlebird *Anthochaera carunculata* eggs, and Van Bael and Pruett-Jones (2000) recorded depredation of Splendid Fairy-wren *Malurus*

splendens nests. However, the species is not well-known as a nest-predator. It is noted that the foraging behaviour and diet of the White-browed Babbler are similar to those of the Eastern Whipbird, identified by Guppy *et al.* (2017) as a primary predator of small bird nests in temperate woodland sites. Furthermore, the congeneric Grey-crowned Babbler *Pomatostomus temporalis* has been identified as a primary woodland nest-predator (Robertson *et al.* 2014), and has also been recorded depredating Splendid Fairy-wren nests (Van Bael and Pruett-Jones 2000).

The unexpected nest-predators identified in this and other studies almost always target eggs rather than hatched young. In general, depredation risk for avian nests is highest in the nestling phase, when increased activity at the nest and begging young are likely to attract the attention of predators (Muchai and du Plessis 2005; Ibáñez-Álamo *et al.* 2012). It is therefore of considerable interest that nests may also be targeted or opportunistically depredated during the laying and incubation phases by species such as the White-browed Babbler and Eastern Whipbird. Although parental investment in the breeding attempt is lower during the laying and incubation phases than in the nestling phase, nest failure and re-nesting are nonetheless costly for the breeding pair (Antczak *et al.* 2009).

The White-browed Babbler is a frequent occupant of grassy woodland restoration plantings (D. Belder unpubl. data). Although there is no strong evidence of this species acting as a primary predator, it is worth considering that the presence of the species in restoration plantings may present a predation risk for small woodland birds breeding in these sites.

This paper presents the first photographic evidence of likely nest-depredation by the White-browed Babbler. Identifying nest-predators is often difficult for several reasons. Firstly,

detailed studies of nest success are undertaken infrequently, perhaps because of the labour-intensive methods required to obtain data on nest success and evidence of predation events. In the absence of empirical evidence, predation risk and primary nest-predators can at best be loosely inferred using pattern data and pre-existing knowledge of species' behaviour. Secondly, studies of nest success have persistently used artificial nest experiments to identify nest-predators, but there is often little or no correspondence between predators of artificial and natural nests (Zanette 2002). Thirdly, unless a chance observation is made, researchers typically rely on remote surveillance via wildlife cameras to capture nest-depredation events. Use of wildlife cameras can be problematic because of high costs (both financial and labour) and low reliability (Newey *et al.* 2015). The example described in the present paper illustrates how difficult it can be to obtain irrefutable evidence of nest-depredation, and subsequently profile unexpected and/or uncommon nest-predators. For example, it is possible that a different camera angle and shorter minimum capture interval might have provided clearer images of this suspected nest-depredation event. Despite these challenges, further research on nest success is essential to enhance our understanding of nest-depredation dynamics and metapopulation processes of birds in temperate woodland habitats and restoration plantings.

Acknowledgements

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APPENDIX B

Breeding success of woodland birds in Australian box-gum grassy woodland restoration plantings

Presented by D. J. Belder at the *24th International Congress for Conservation Biology* in Cartagena, Colombia, 21-25 July 2017.

Abstract

Box-gum grassy woodlands are among Australia's most threatened ecological communities, due largely to land conversion for agriculture. Many bird species associated with these woodlands are suffering population declines due to habitat loss and fragmentation. Restoration plantings aim to increase habitat quality and connectivity for native wildlife in these highly fragmented agricultural landscapes.

Our research investigates bird breeding success in box-gum grassy woodland restoration plantings on farms in south-eastern Australia. The primary aim is to assess the habitat quality of plantings for woodland birds. Habitat quality is typically inferred through pattern data, such as species distribution and abundance. However, it is important to consider population dynamics such as breeding success when evaluating habitat quality and assessing whether restoration plantings are meeting conservation goals.

To determine whether restoration plantings provide suitable habitat for supporting resident woodland bird populations, we examined how breeding activity and daily nest success in plantings compared to that in remnant woodland patches, and explored the effects of different planting characteristics (e.g. size and shape) on these variables. We monitored over 250 nests

across 21 woodland sites over 2 breeding seasons. Average nest success across all sites was close to the typical rate of around 30%. However, our results indicate much lower success rates and higher incidence of nest predation in remnant woodland sites compared with restoration plantings. Smaller plantings appear to provide the highest quality breeding habitat for woodland birds, with fledging rates four times higher than in similar-sized remnants, and twice as high as in large reference sites. Our findings suggest that restoration plantings have the potential to match and even surpass remnant woodland patches in providing quality habitat for woodland birds in a fragmented box-gum grassy woodland landscape.

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APPENDIX C

Is bigger always better? Influence of restoration planting size on woodland bird breeding activity

Presented by D. J. Belder at the *2018 Annual Conference of the Ecological Society of Australia* in Brisbane, Queensland, 25-29 November 2018.

Abstract

Restoration plantings are an increasingly common way to address habitat loss in agricultural landscapes. Native fauna, including birds, may readily occupy planted areas of vegetation. However, unless restoration plantings support breeding populations, their effectiveness as a conservation strategy may be limited. We assessed breeding activity of woodland birds in restoration plantings in the South-west Slopes bioregion of New South Wales, Australia. We compared breeding activity in plantings of different size (small and large) and shape (linear and block-shaped), and in remnant woodland sites. Contrary to expectations, we found that for the woodland bird assemblage, breeding activity increased with decreasing patch size. We found no effect of patch type or shape. Breeding activity increased with decreasing patch size for dome-nesters, but not for cup-nesters. For species of conservation concern, there was no effect of patch type, size, or shape on breeding activity. Our results highlight the value of small habitat patches in fragmented agricultural landscapes, and indicate that restoration plantings are as valuable as remnant woodland patches for supporting woodland bird breeding activity. We also demonstrate the importance of conducting bird breeding studies to assess the conservation value of restoration plantings and other habitat patches for avifauna.

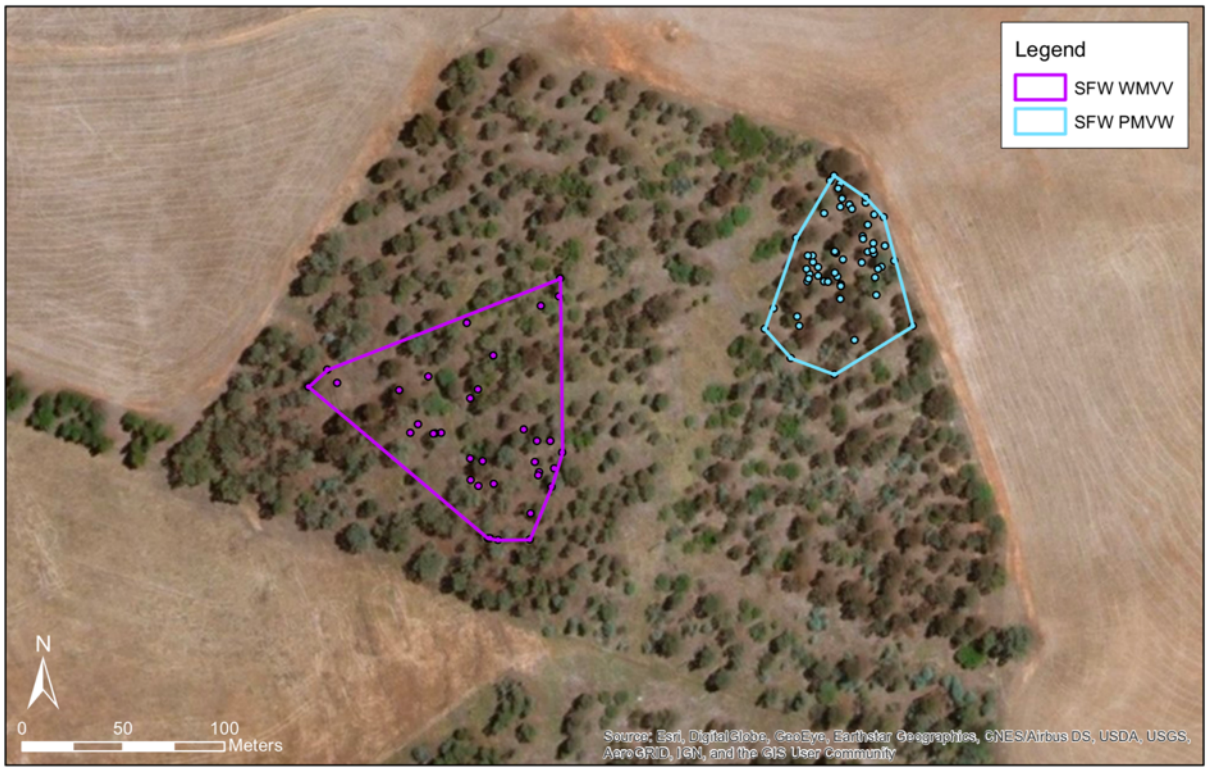
APPENDIX D

Home ranges of superb fairywren (*Malurus cyaneus*) and willie wagtail (*Rhipidura leucophrys*) in restoration plantings and woodland remnants in the South-west Slopes bioregion, NSW

Colour-banded male superb fairywren in a restoration planting. Photo: Madaline Hill.

Superb fairywren home ranges

Block-shaped sites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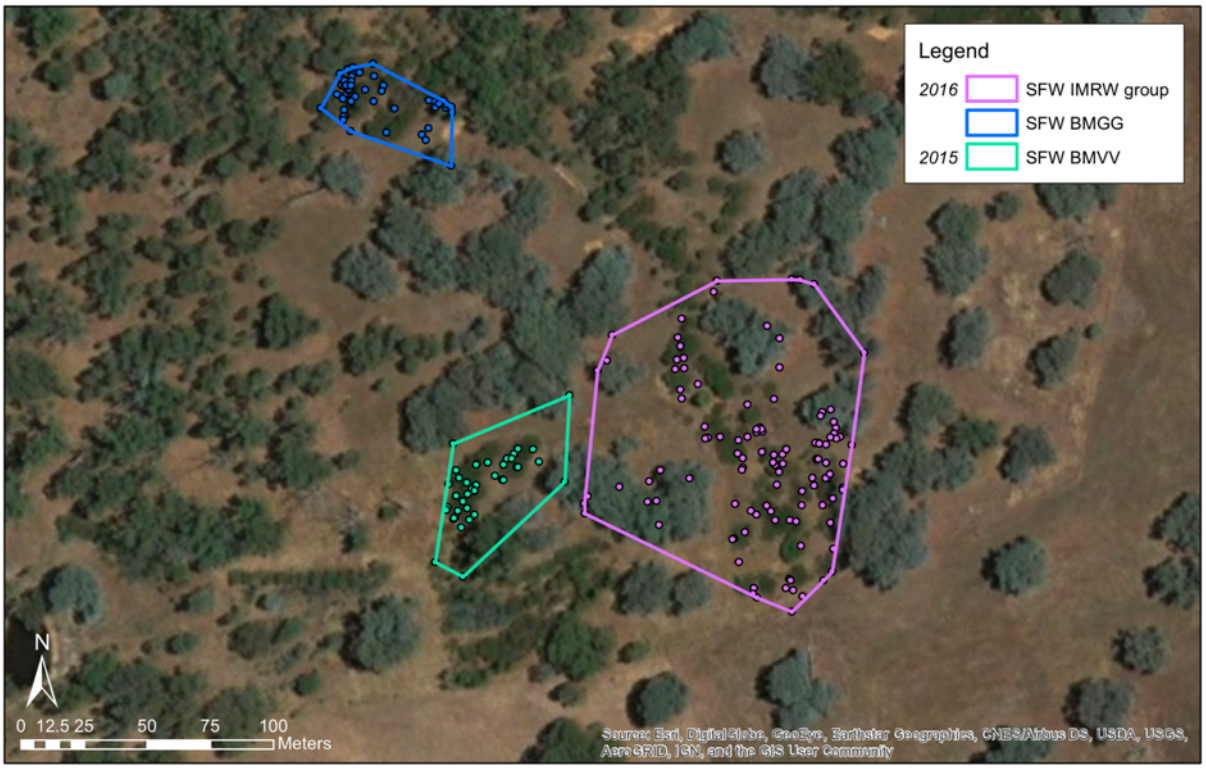
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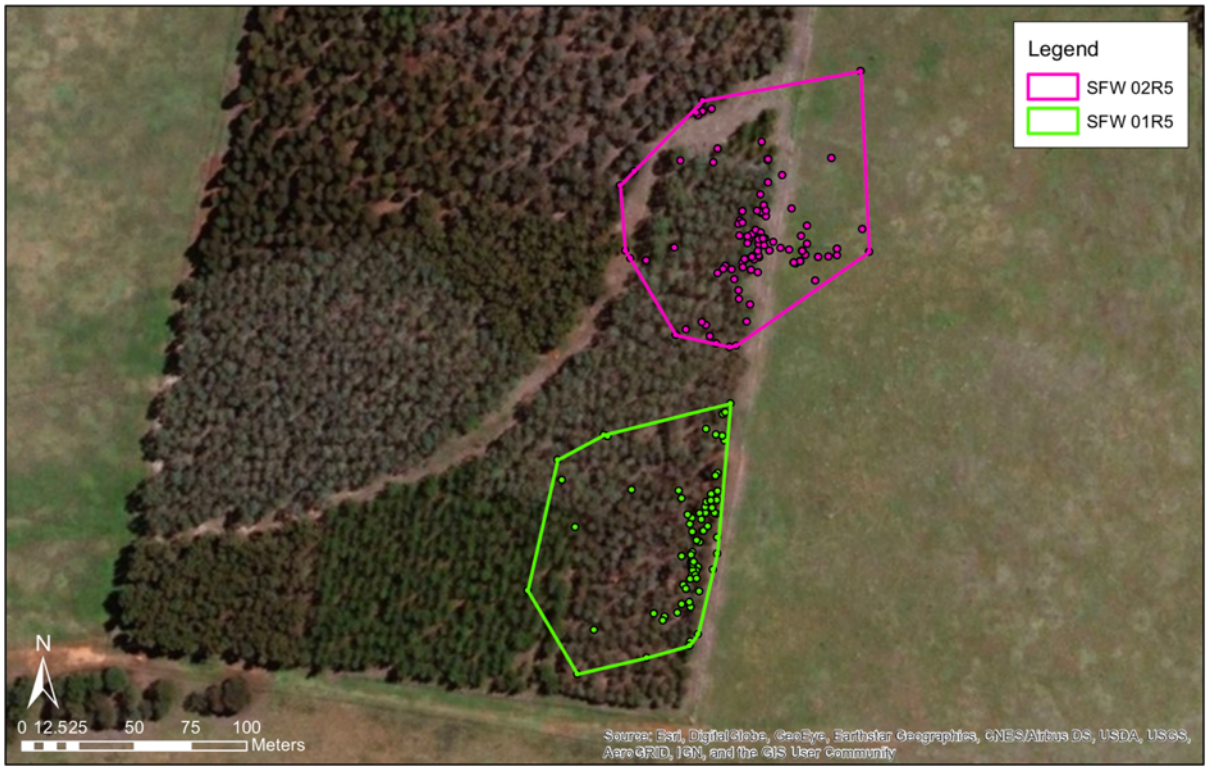
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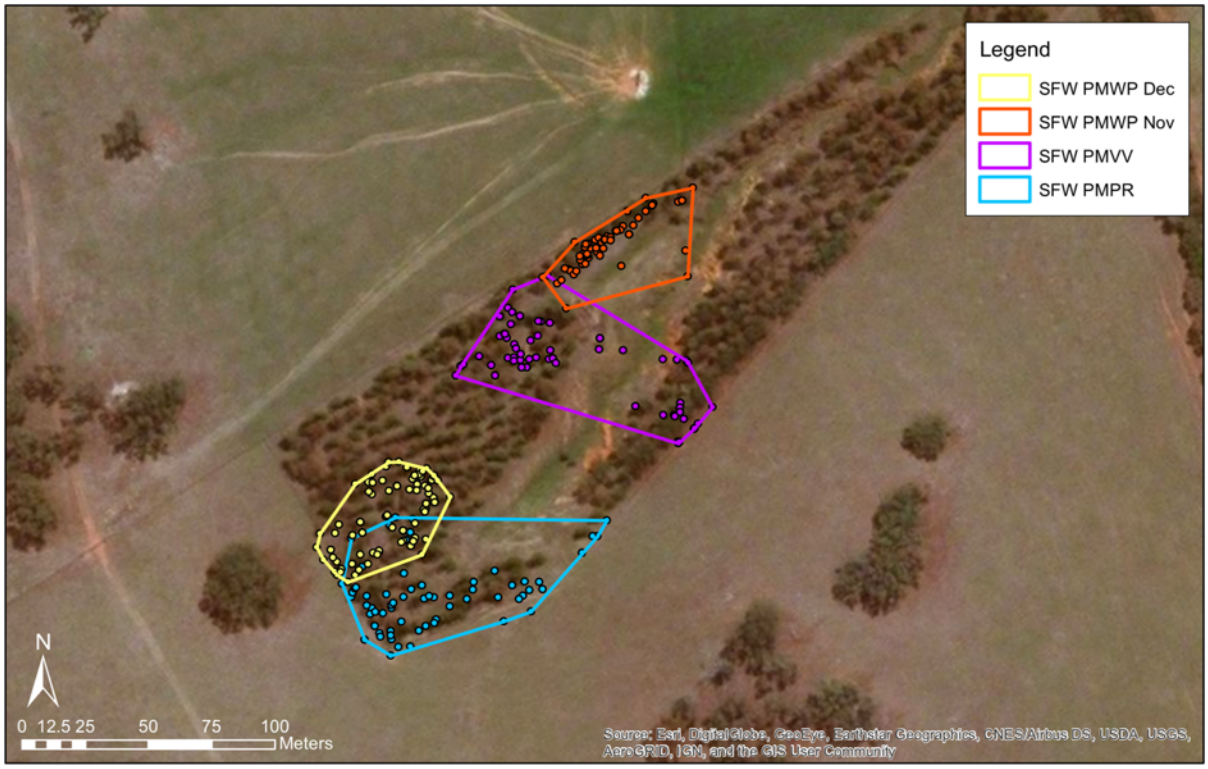
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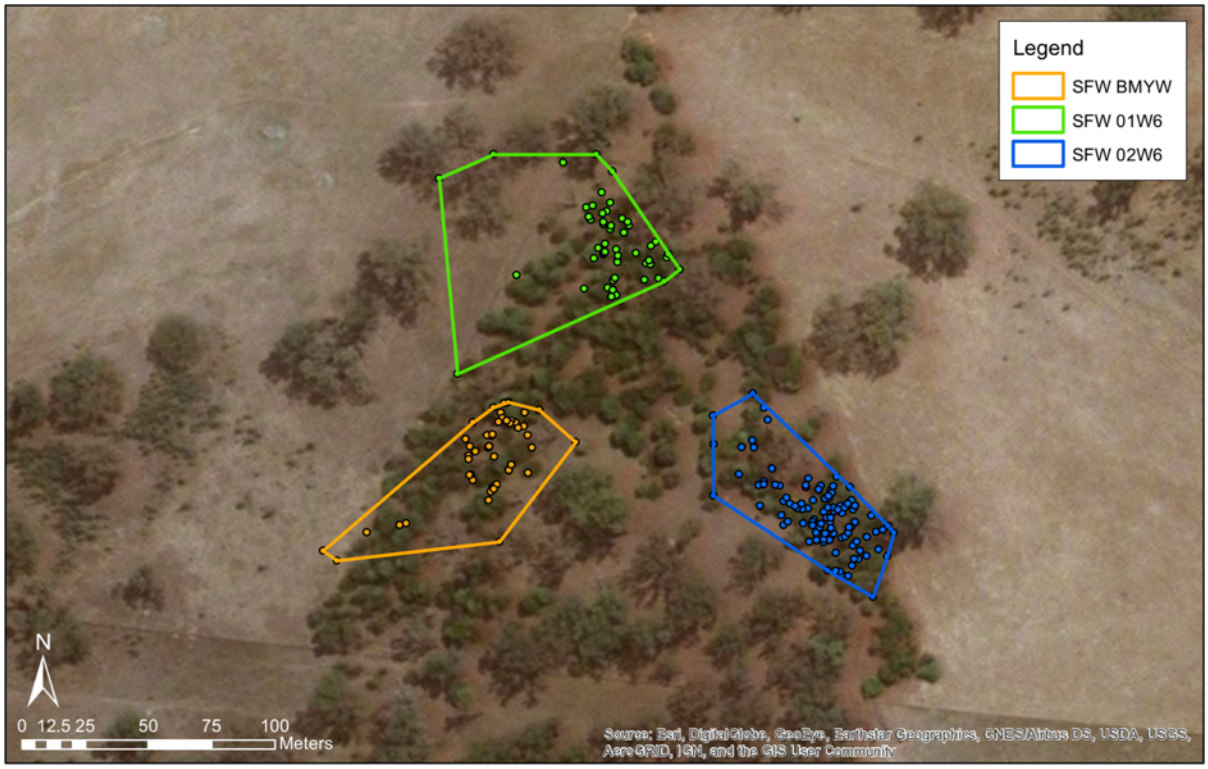
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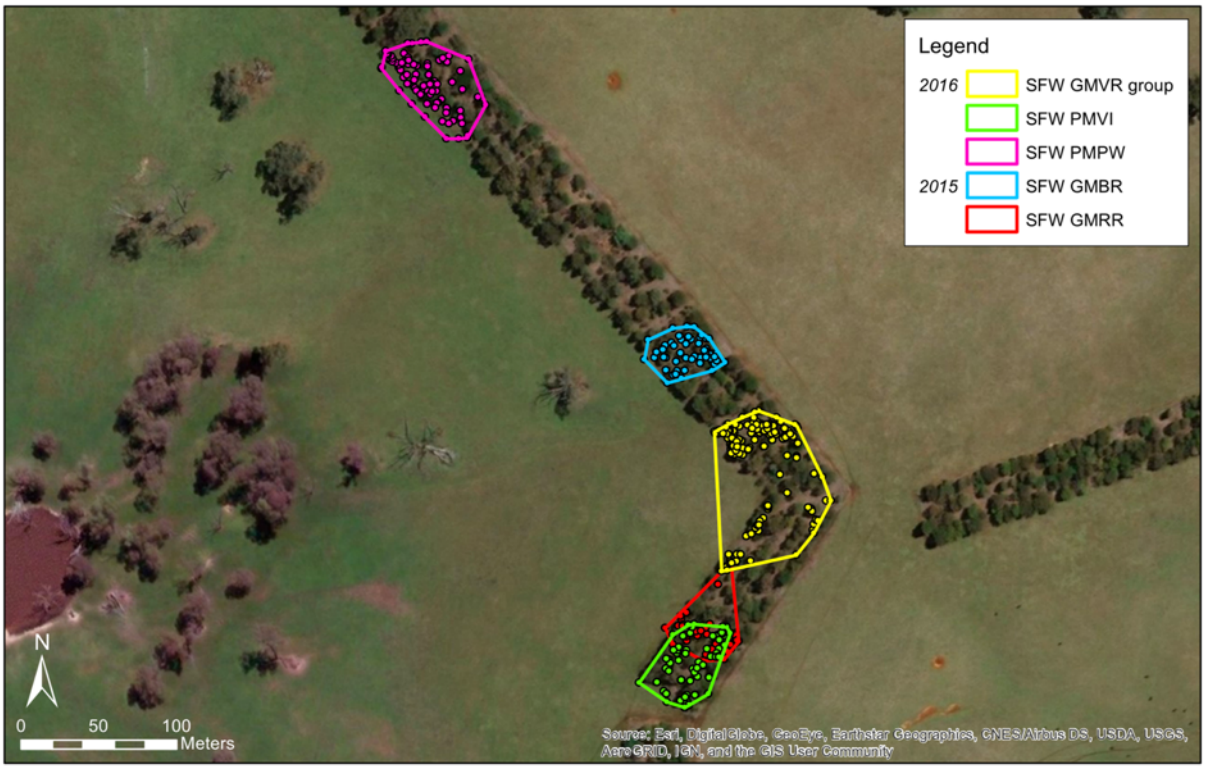
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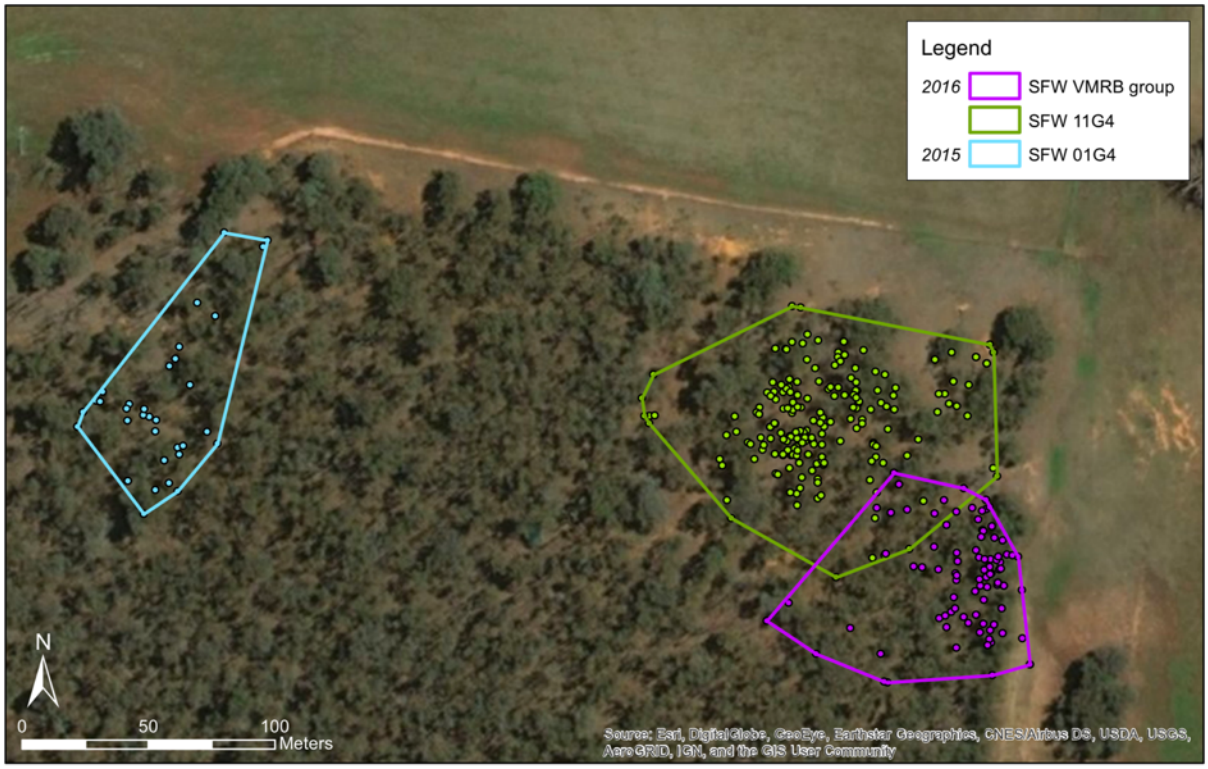
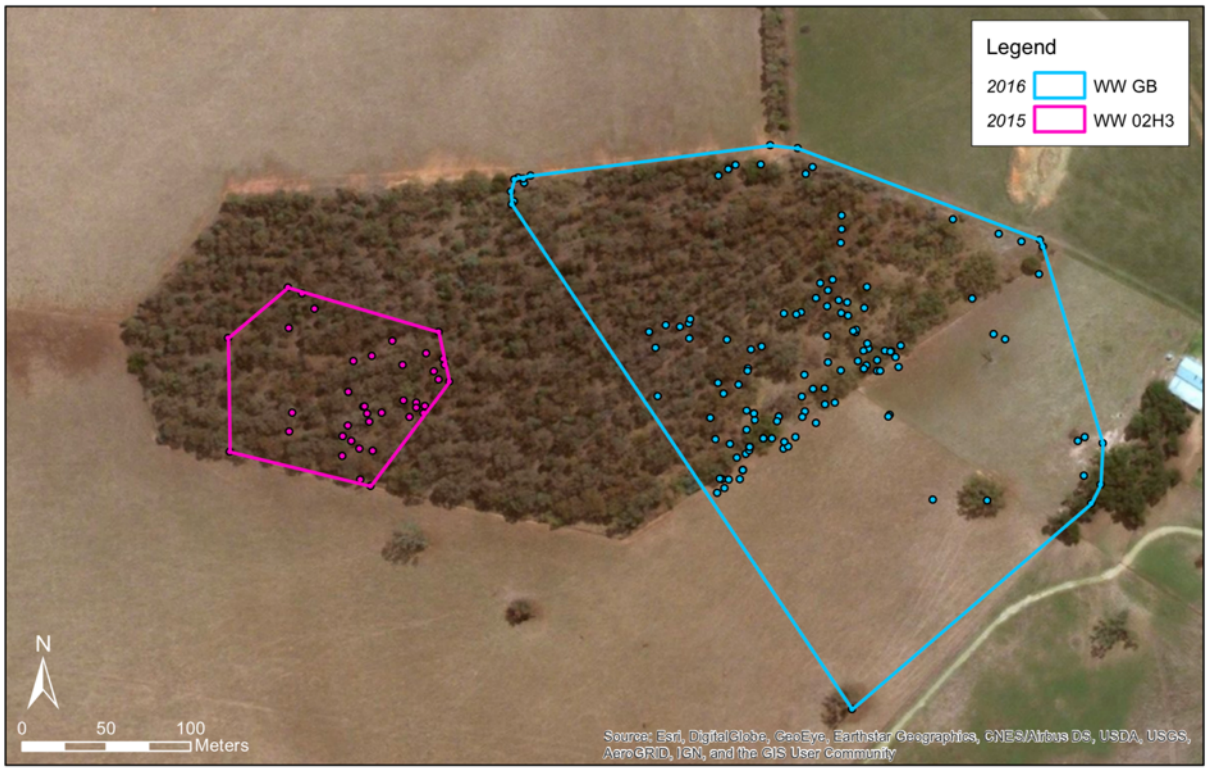


Willie wagtail home ranges and movement patterns

Block-shaped sites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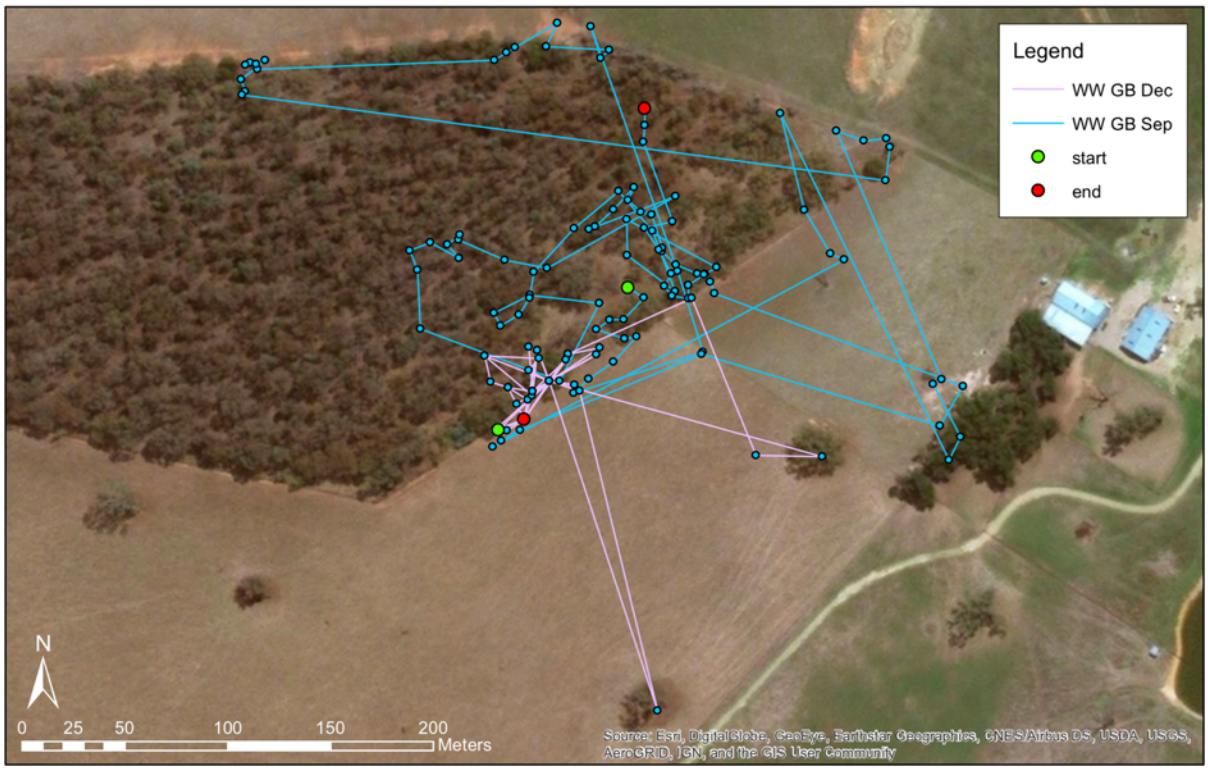
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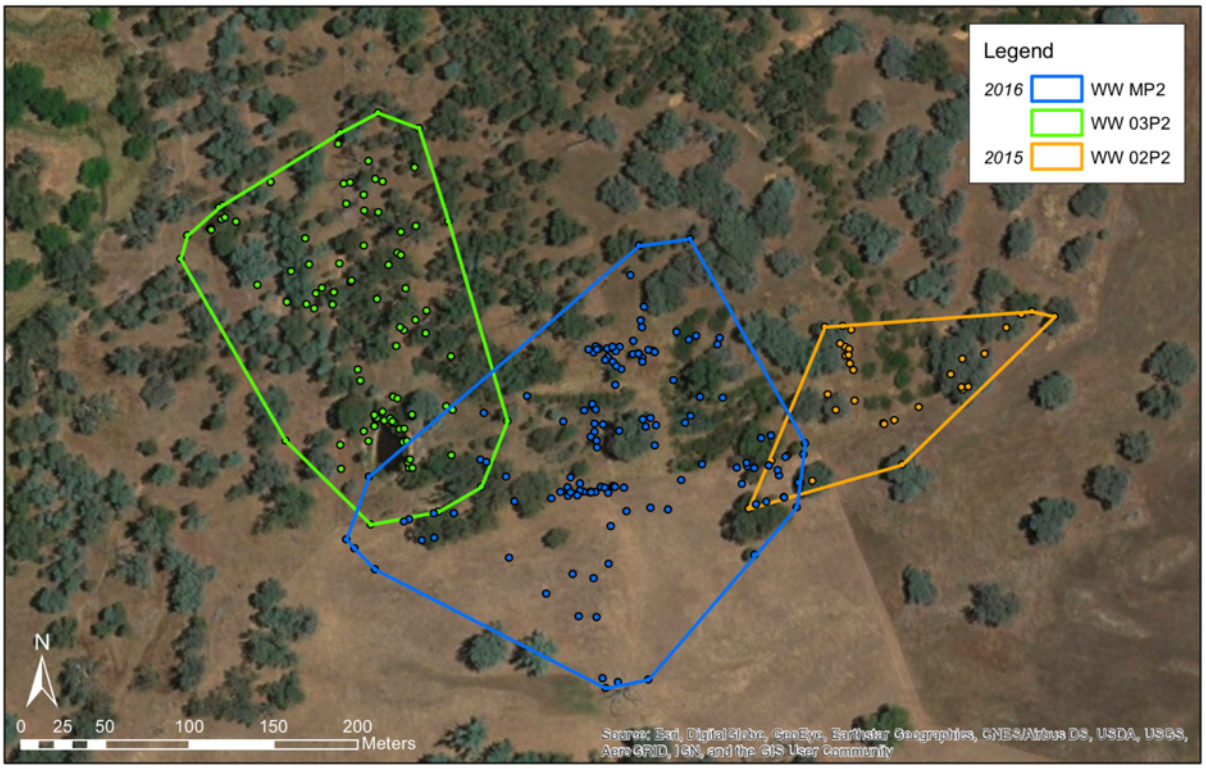
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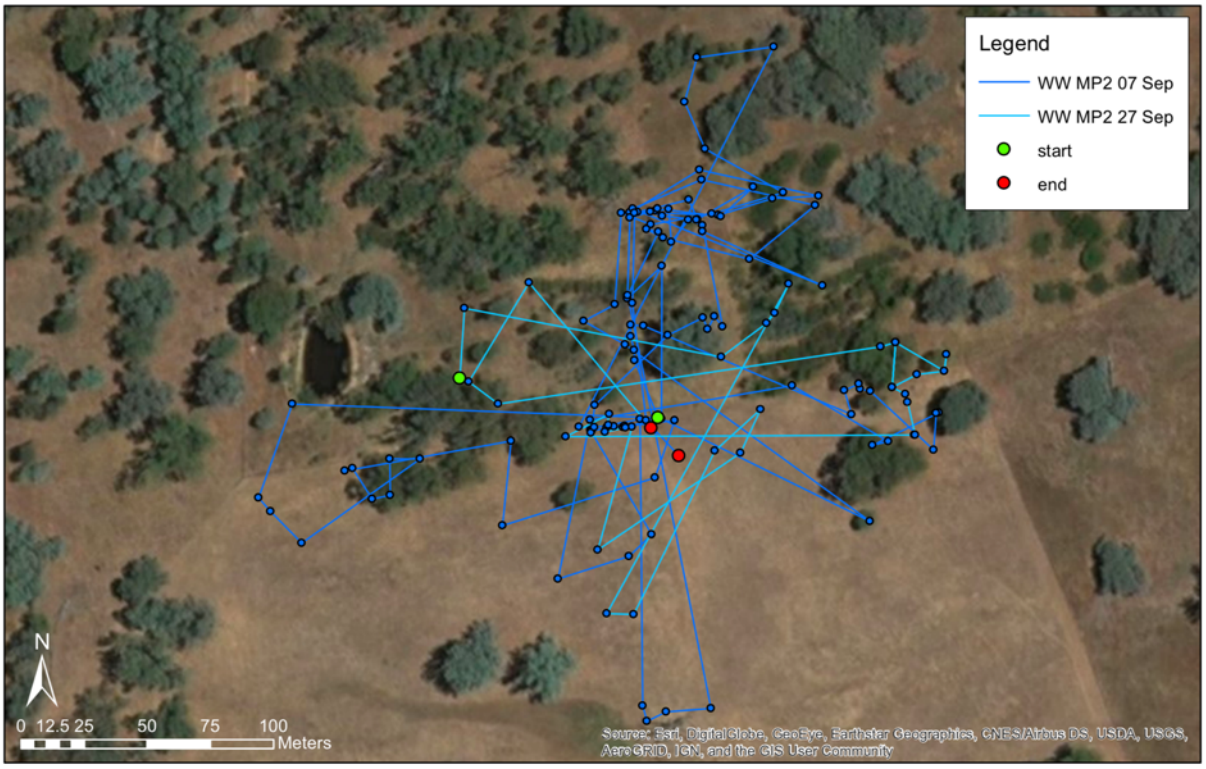
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW MP2 07 Sep
WW MP2 27 Sep
start
end
N
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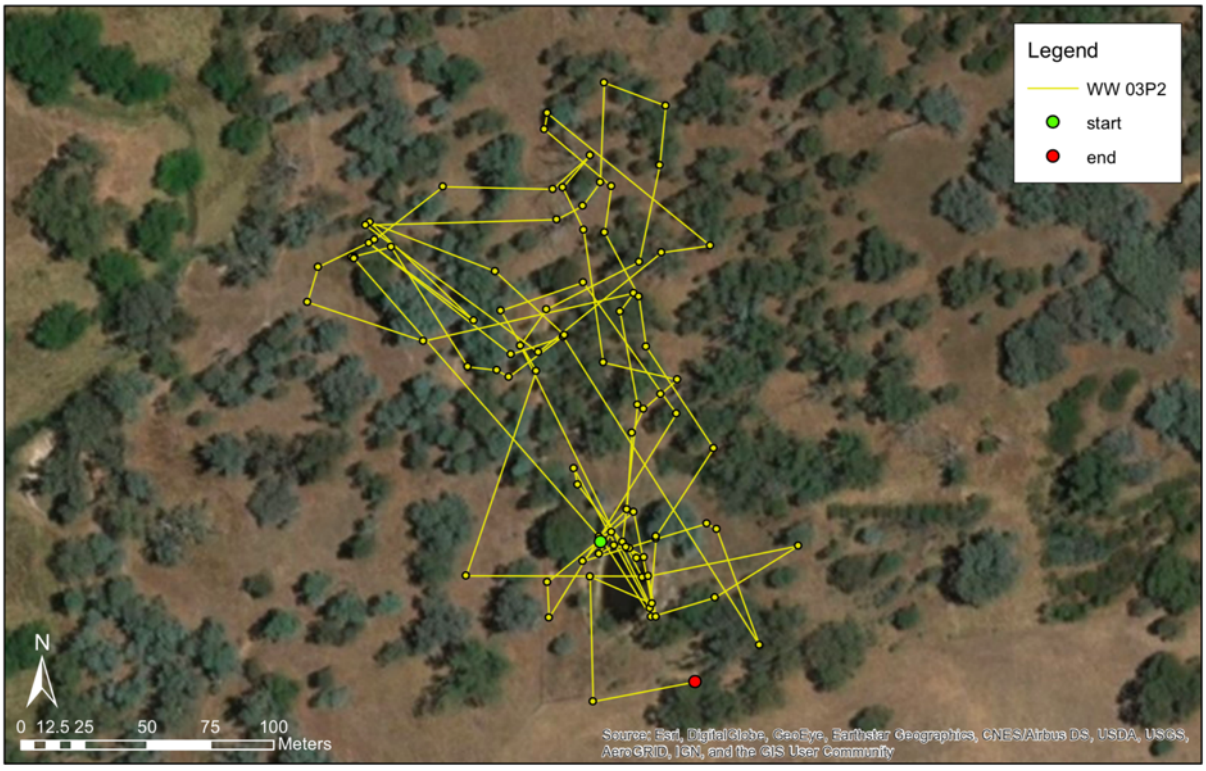
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 03P2
start
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Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
2016 WW 11S5
2015 WW 01S5
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 11S5
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01S5
start
end
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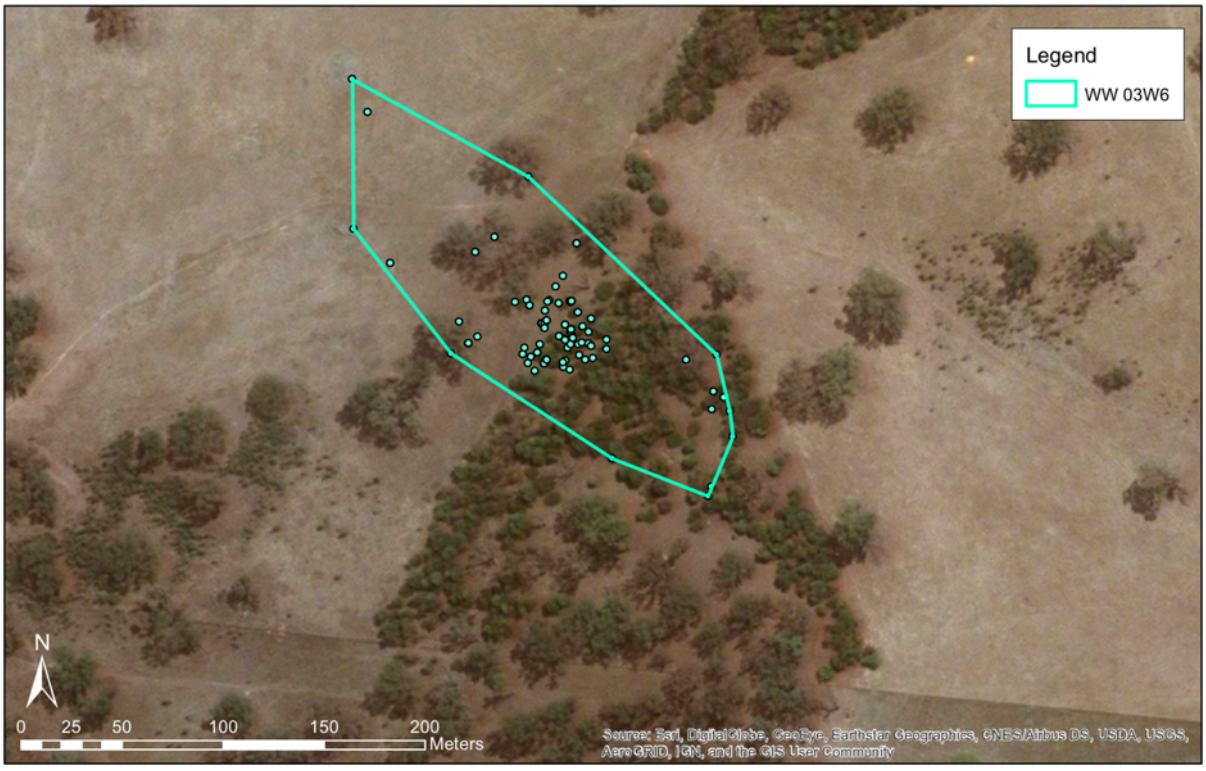
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 03W6
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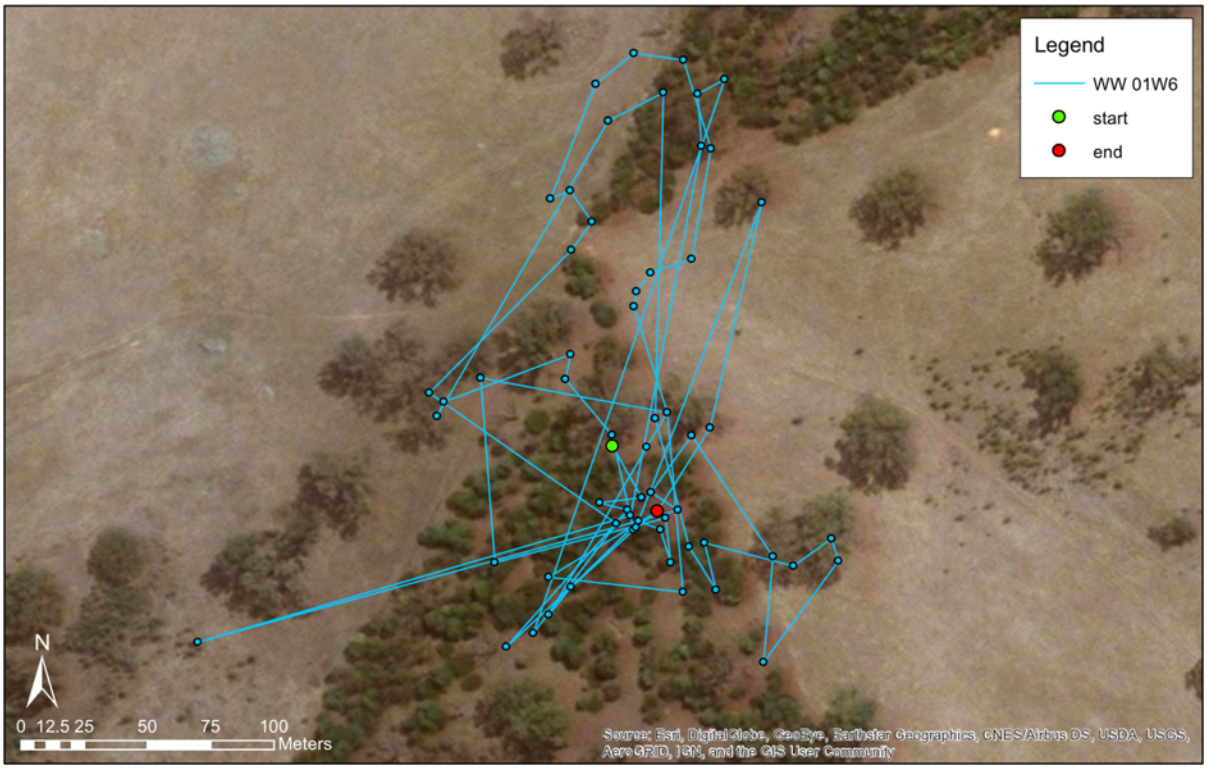
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01W6
start
end
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Meters
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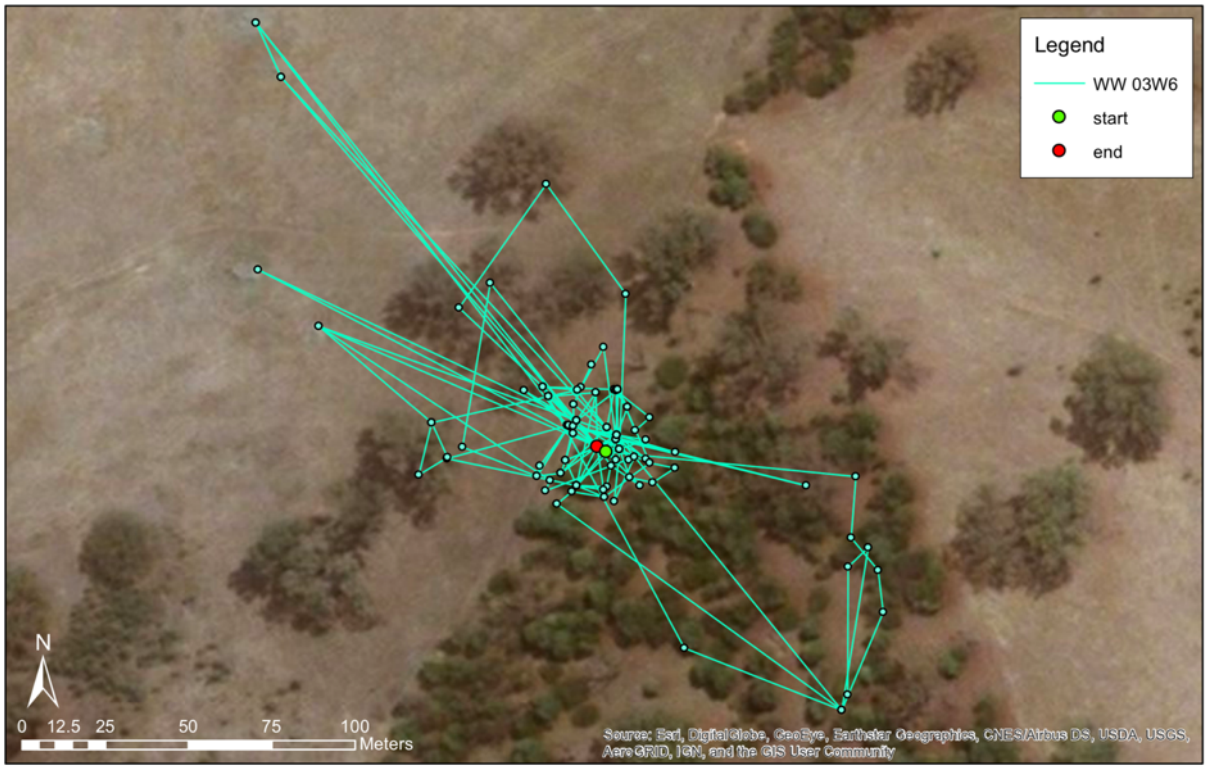
Legend
WW 03W6
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01R5
WW 02R5
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01W3
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Linear sites


Legend
2016 WW MFA
2015 WW 01FA
N
0 50 100 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01FA
start
end
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW MFA
start
end
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW YB
WW 02M3
N
0 50 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01M3
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW YB
start
end
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 02M3
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW BB
N
0 50 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01M6
start
end
N
0 12.525 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW BB
start
end
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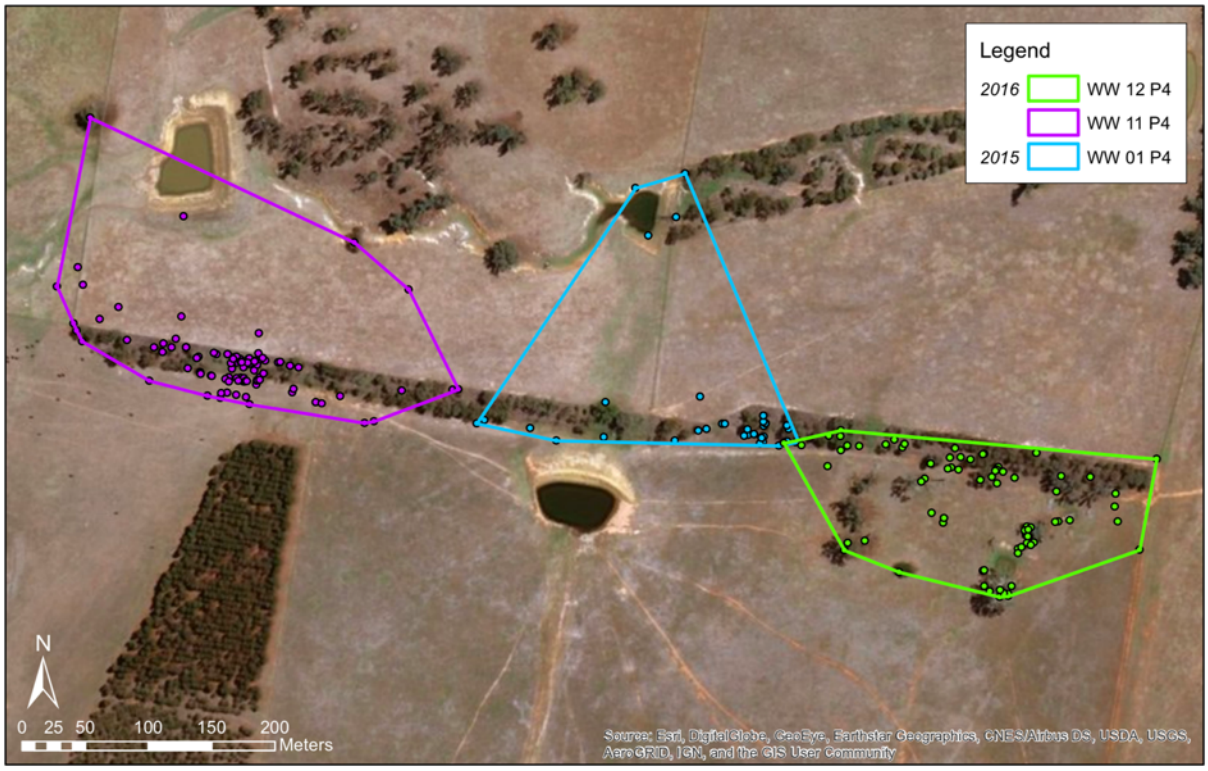
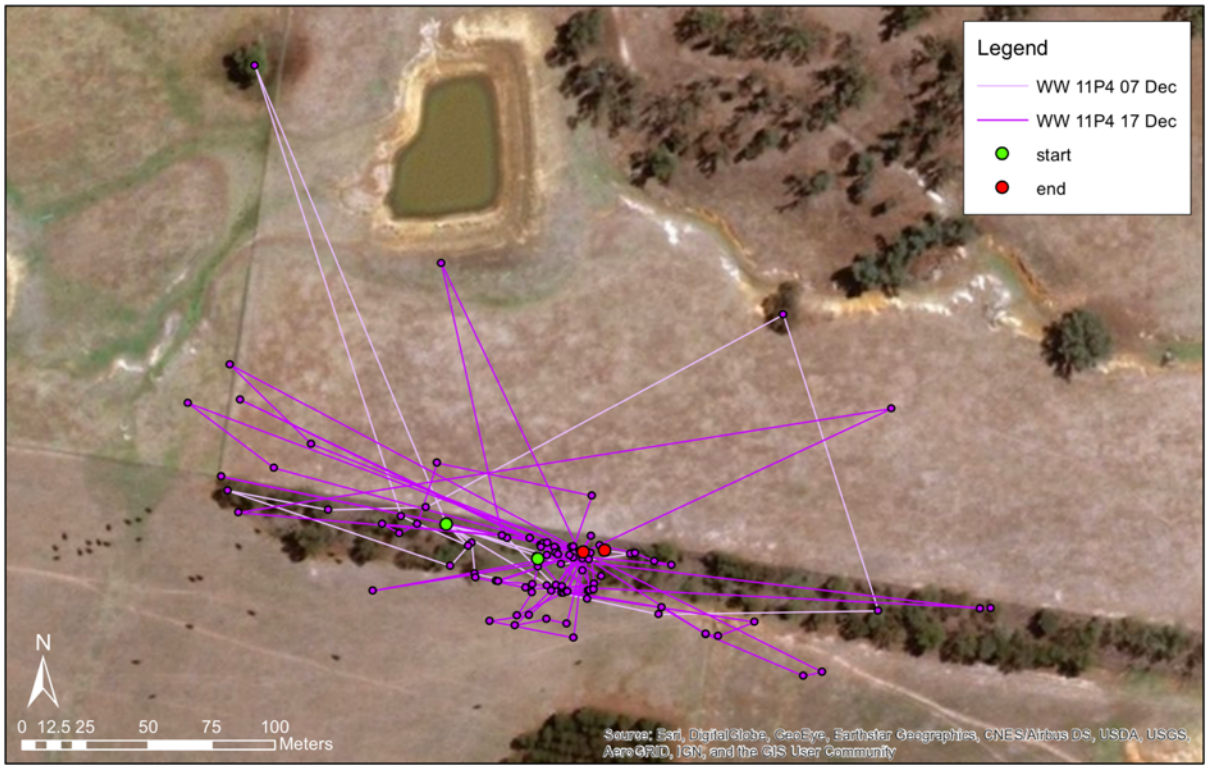
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
2016
WW 12 P4
WW 11 P4
2015
WW 01 P4
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 11P4 07 Dec
WW 11P4 17 Dec
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 01P4
start
end
N
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Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 12P4 13 Jun
WW 12P4 15 Jun
start
end
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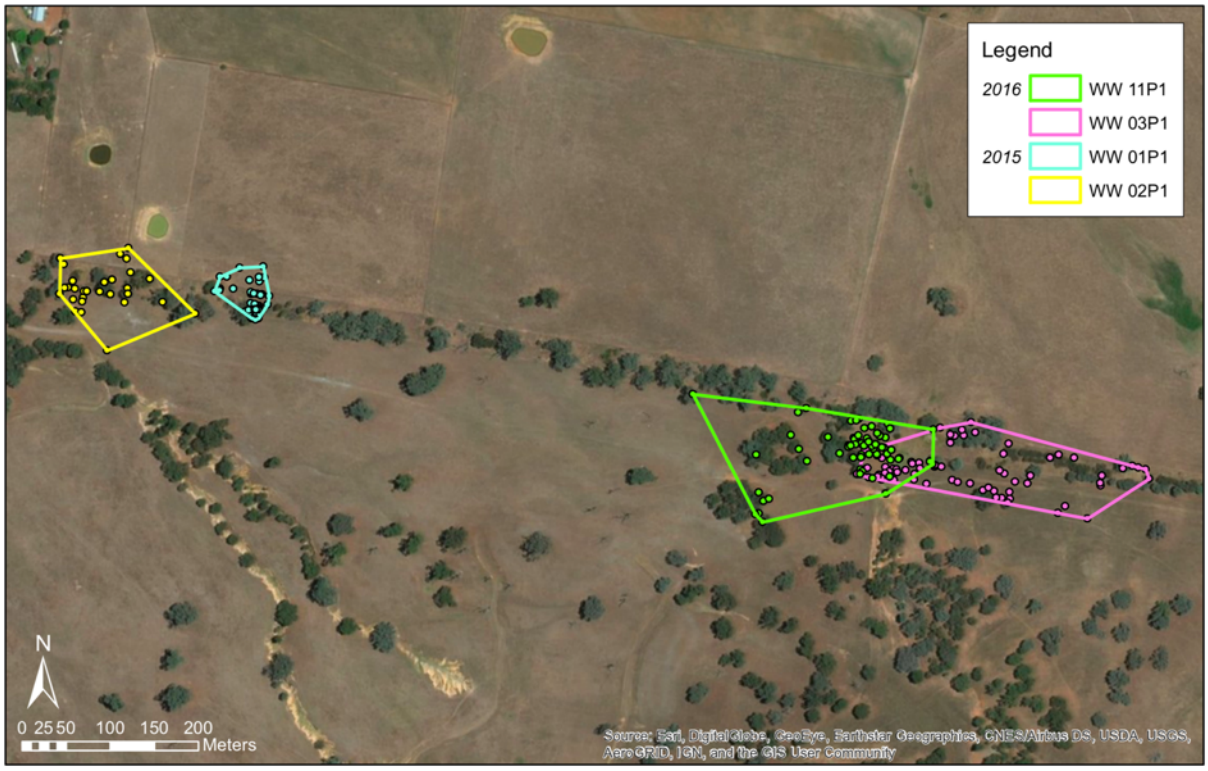
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
2016
WW 11P1
WW 03P1
2015
WW 01P1
WW 02P1
N
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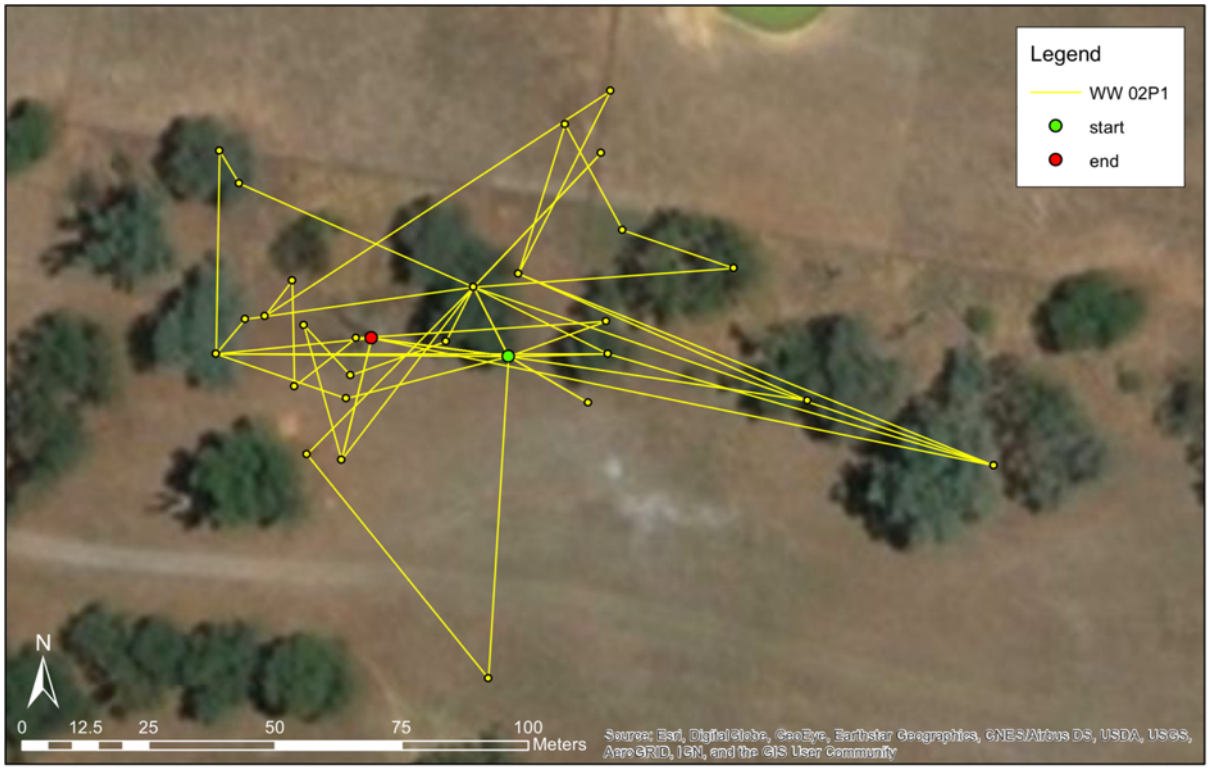
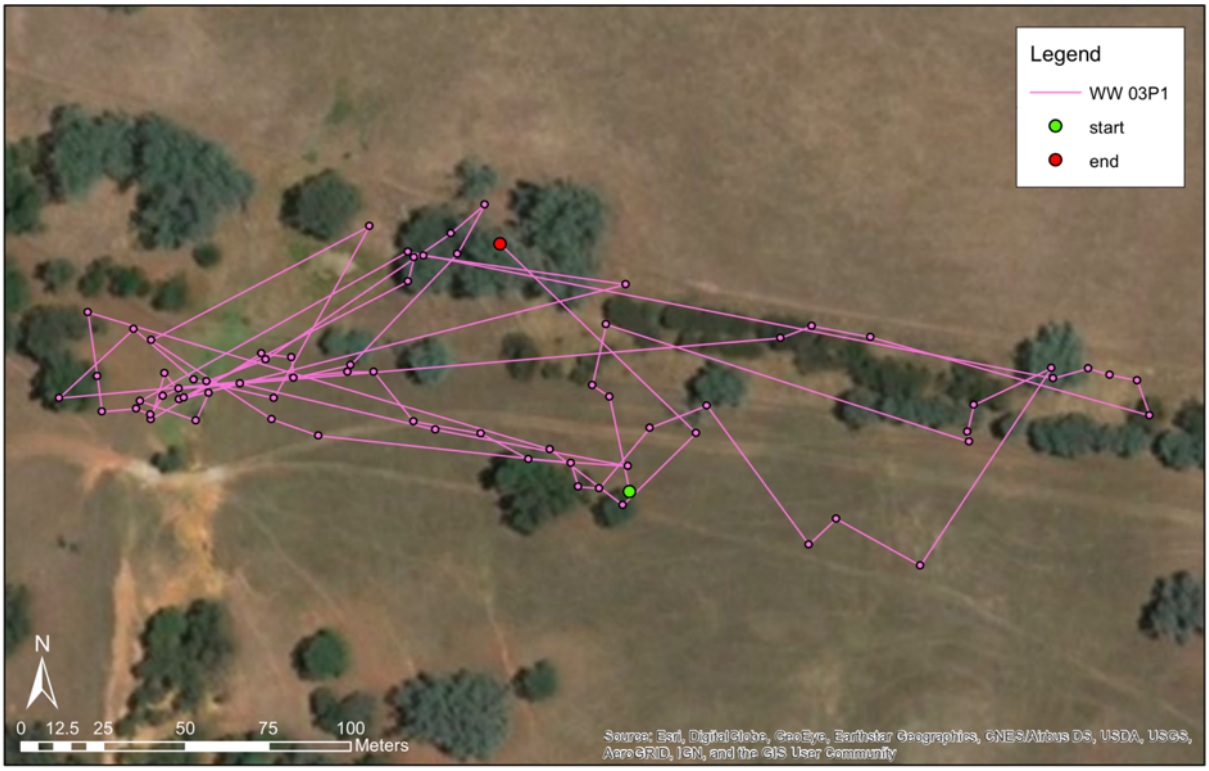
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 02P1
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 11P1
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 03P1
start
end
N
0 12.5 25 50 75 100
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW MSC1
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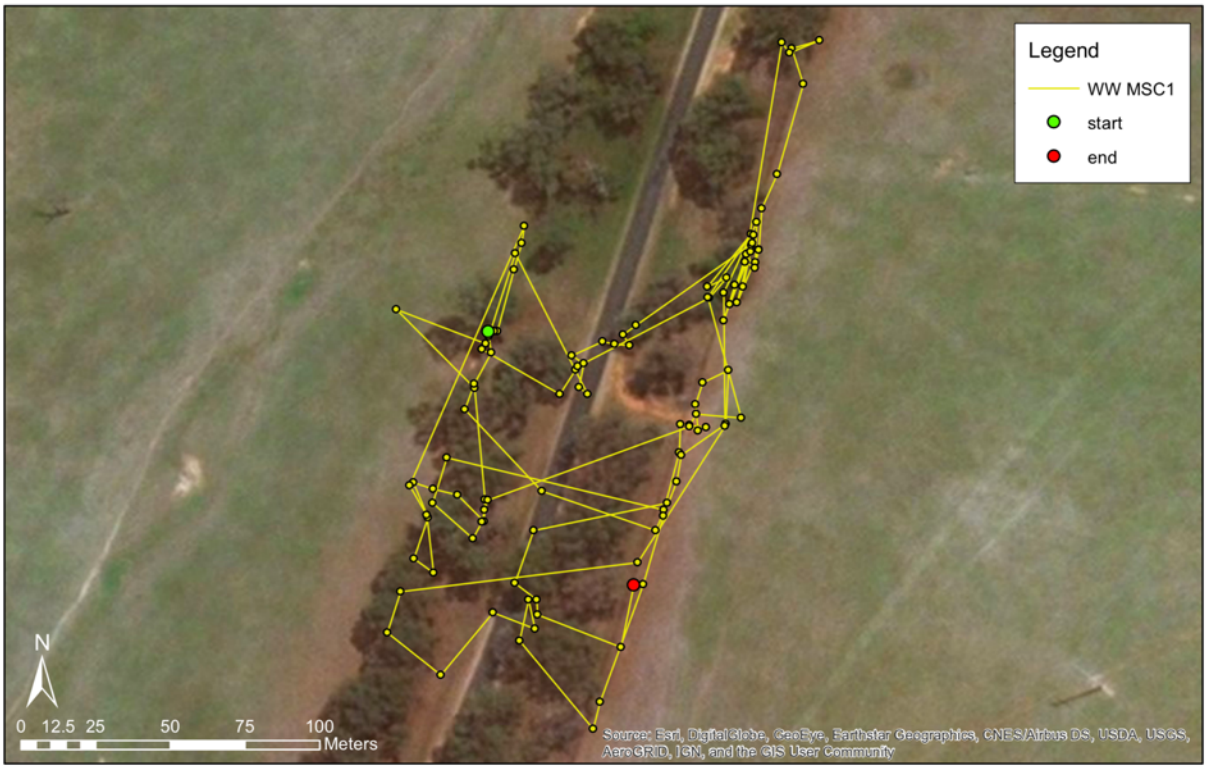
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW MSC1
start
end
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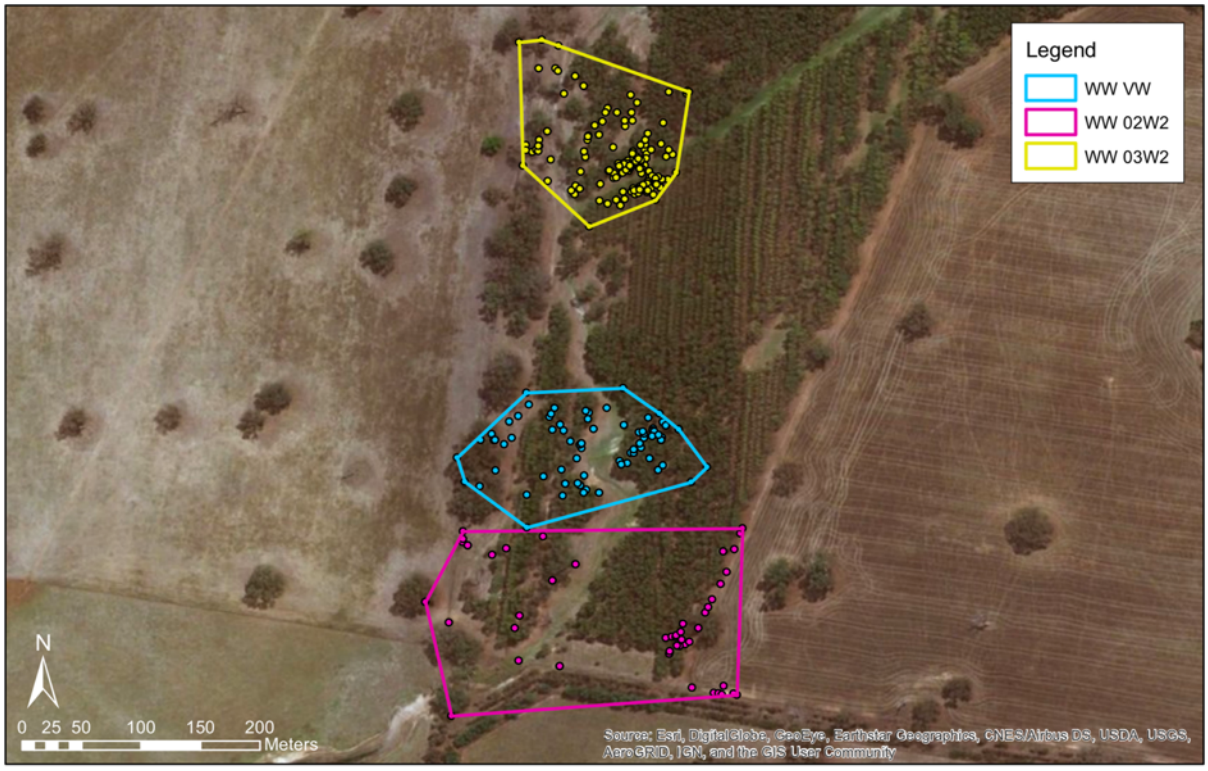
Legend
WW VW
WW 02W2
WW 03W2
N
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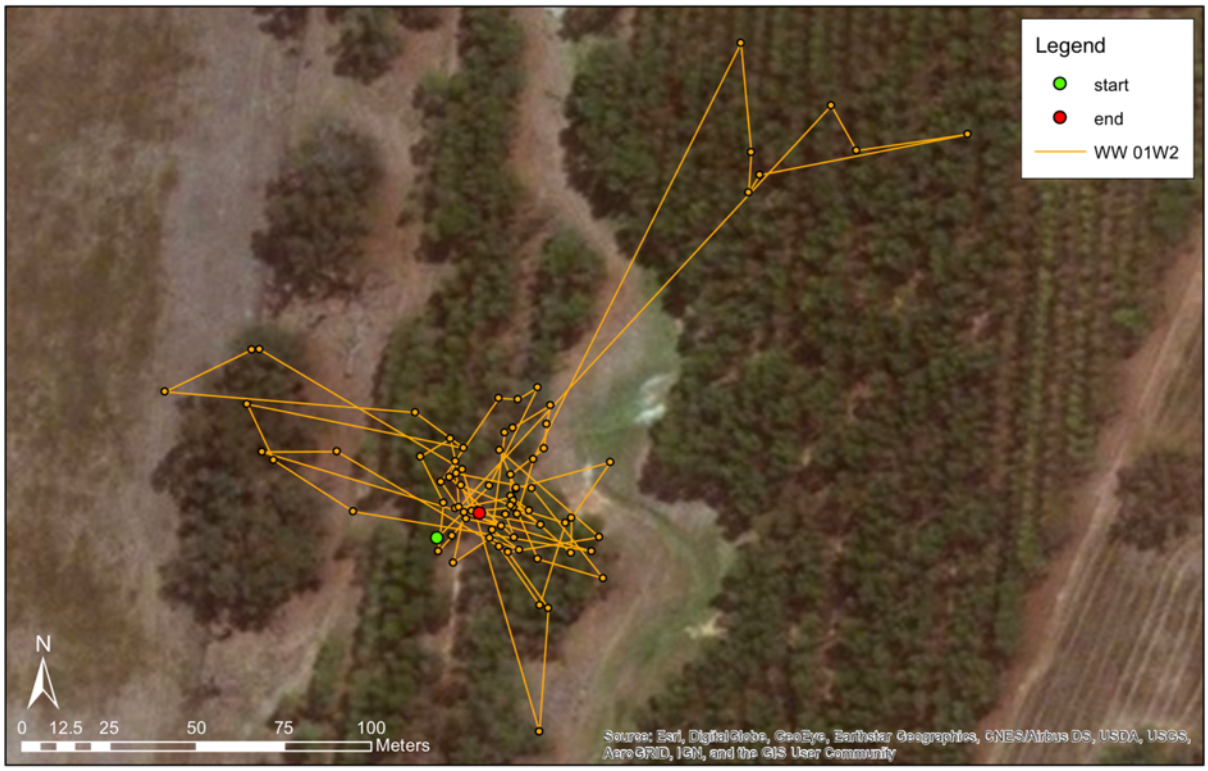
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
start
end
WW 01W2
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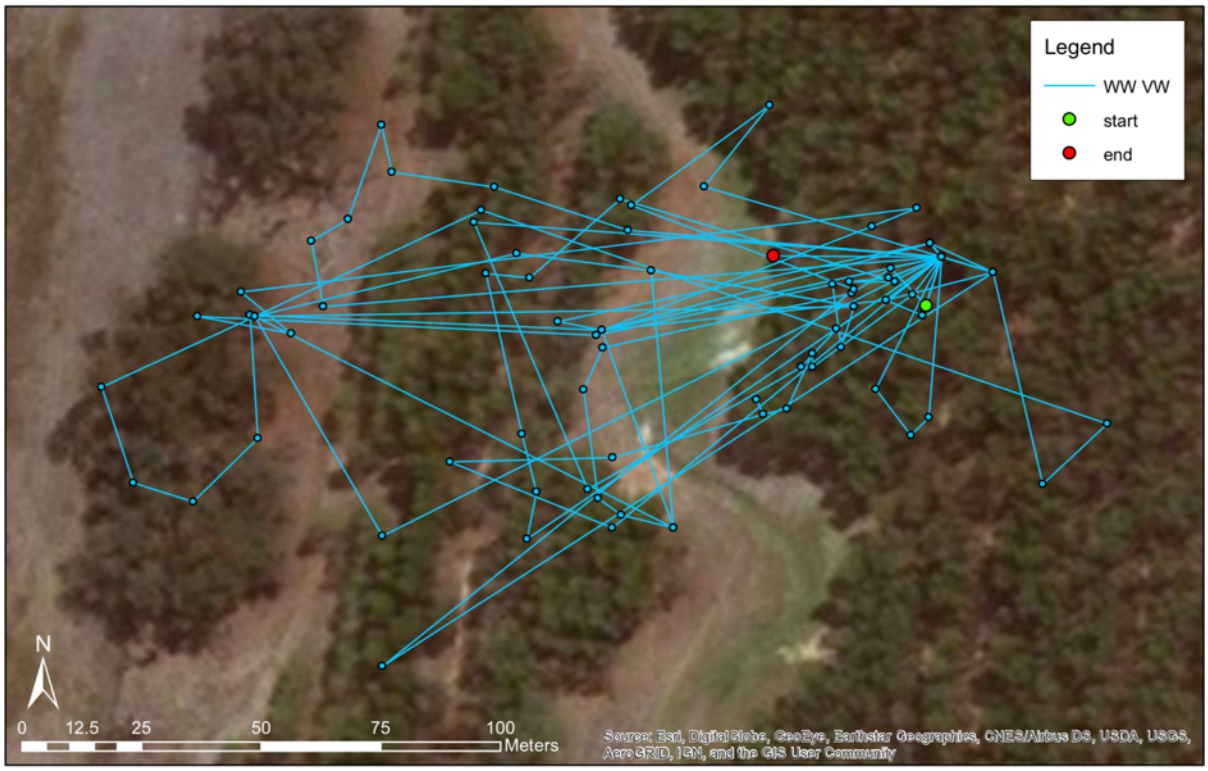
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW VW
start
end
N
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Meters
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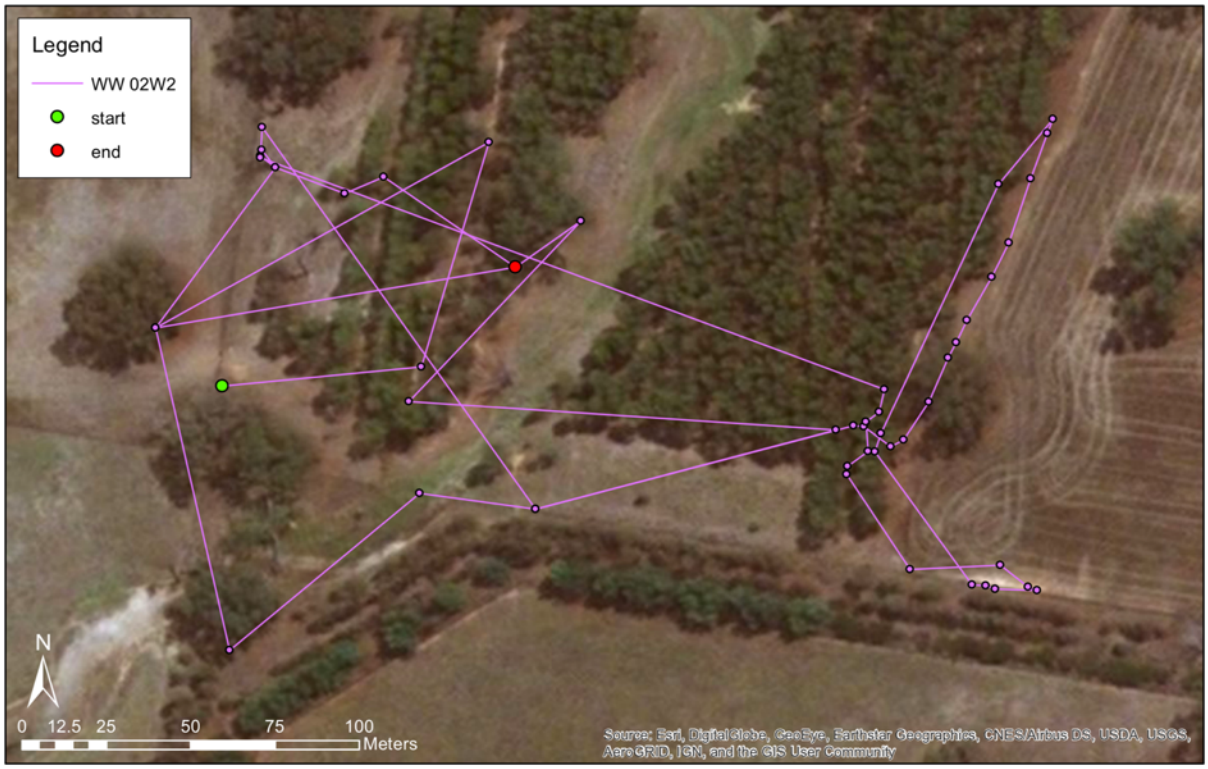
Legend
WW 02W2
start
end
N
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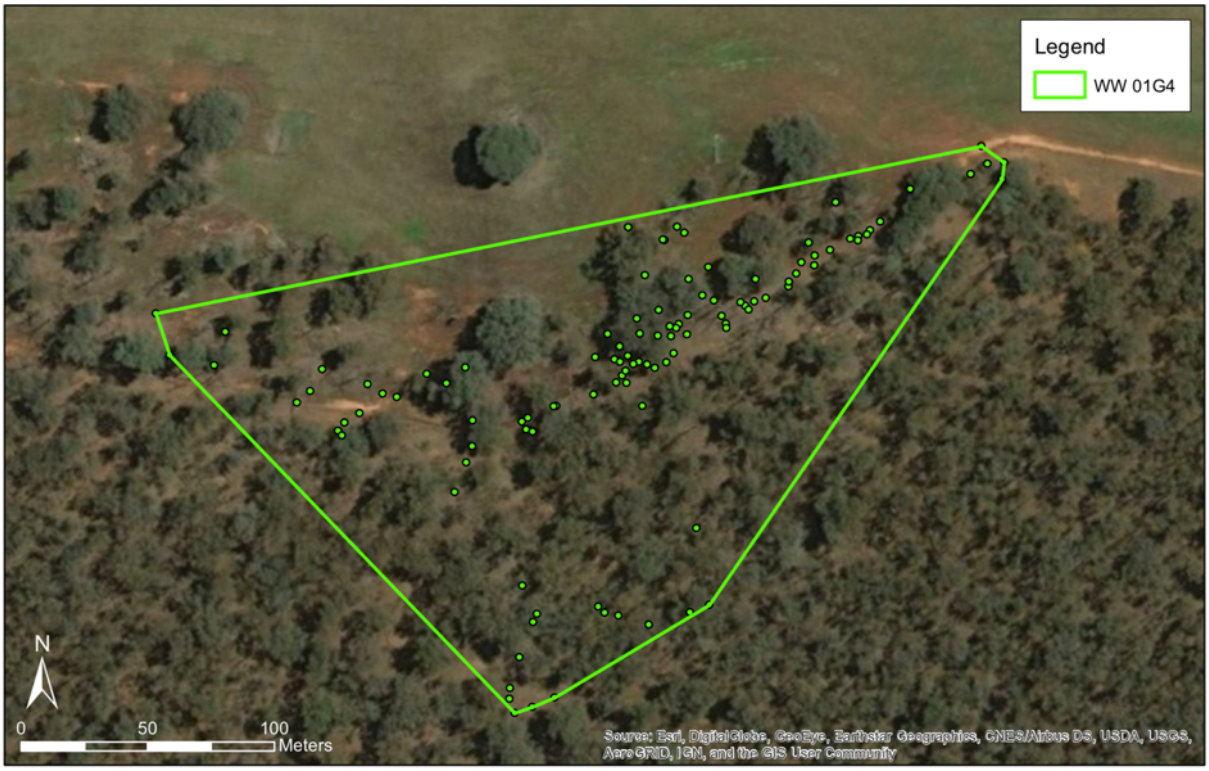
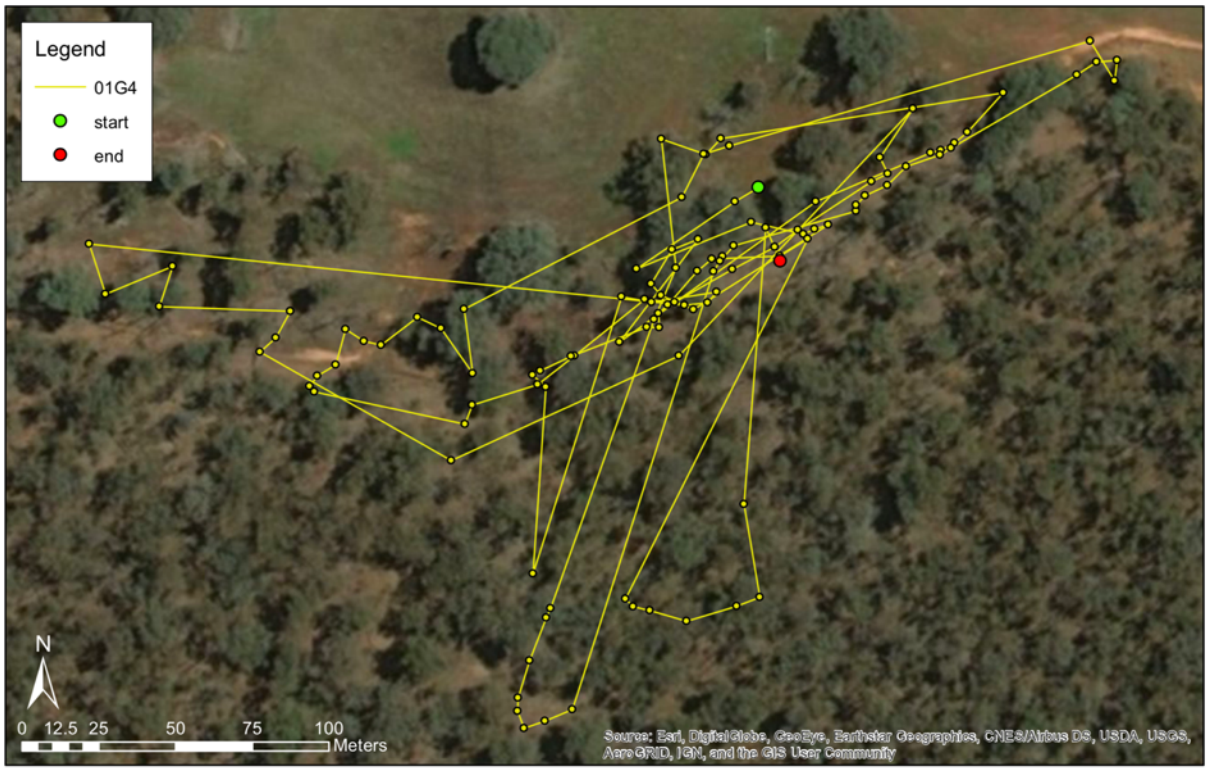
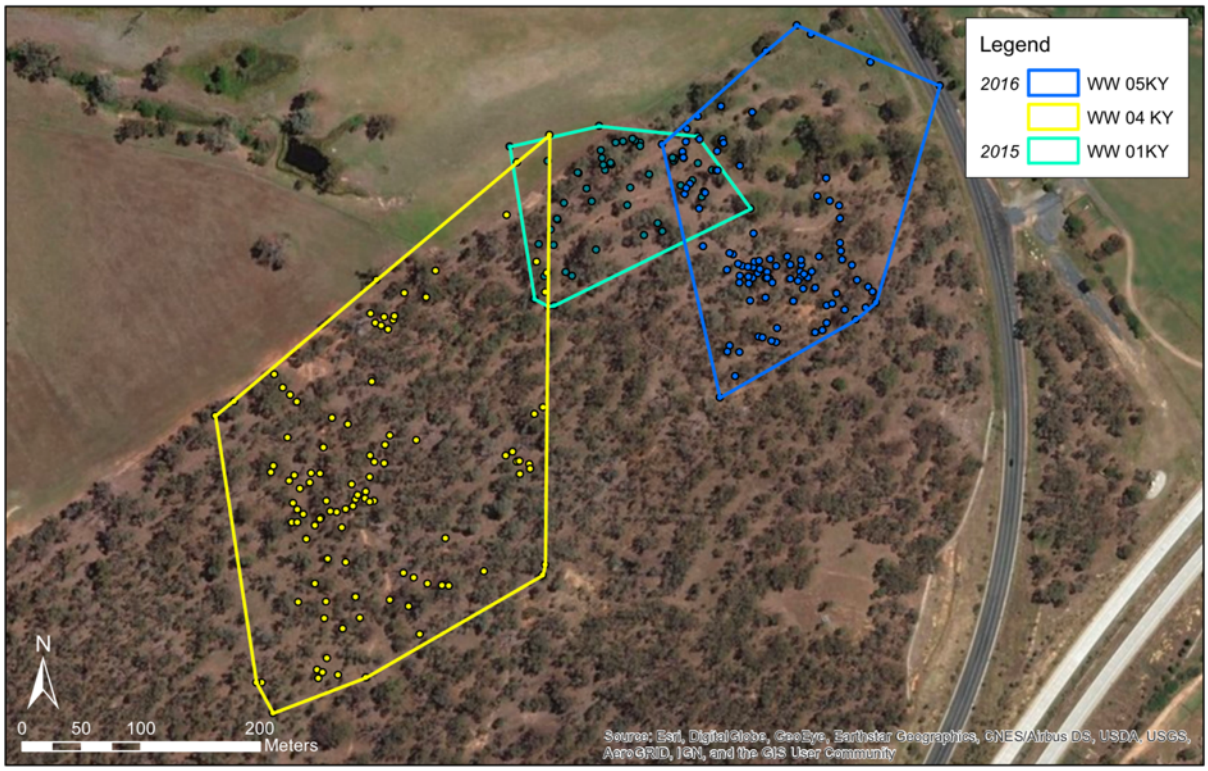
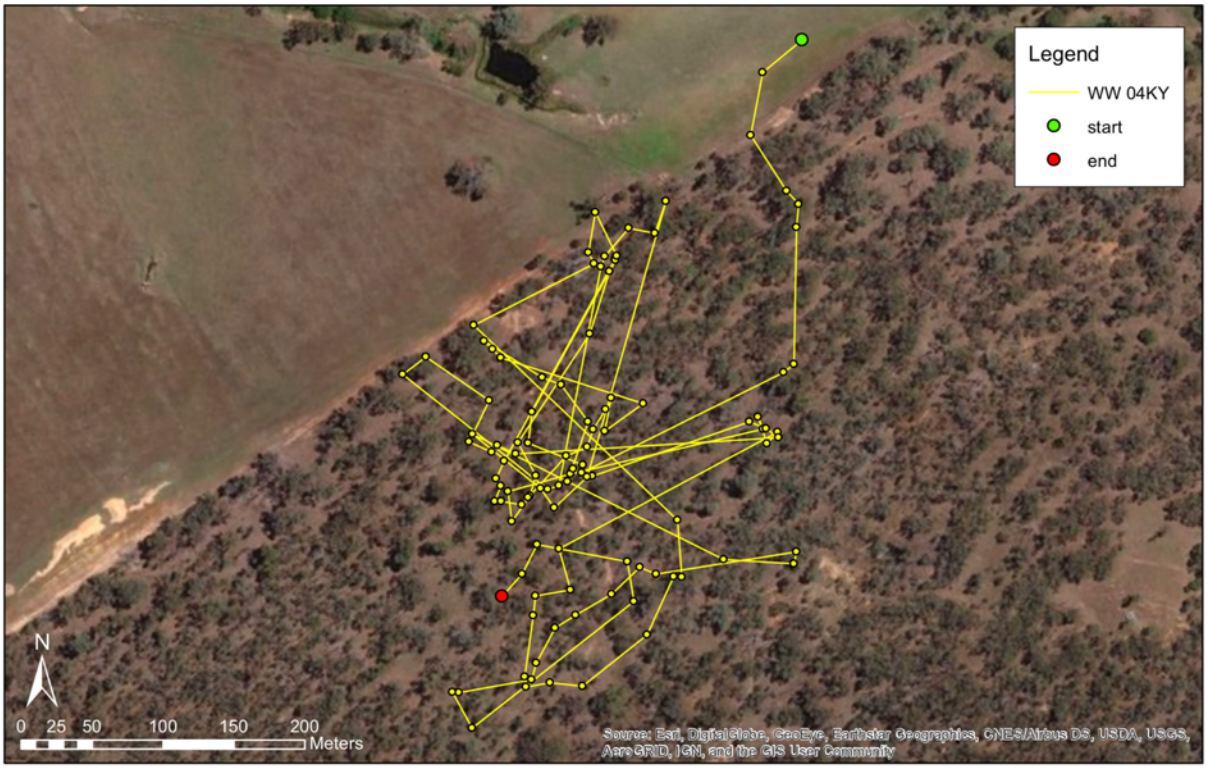
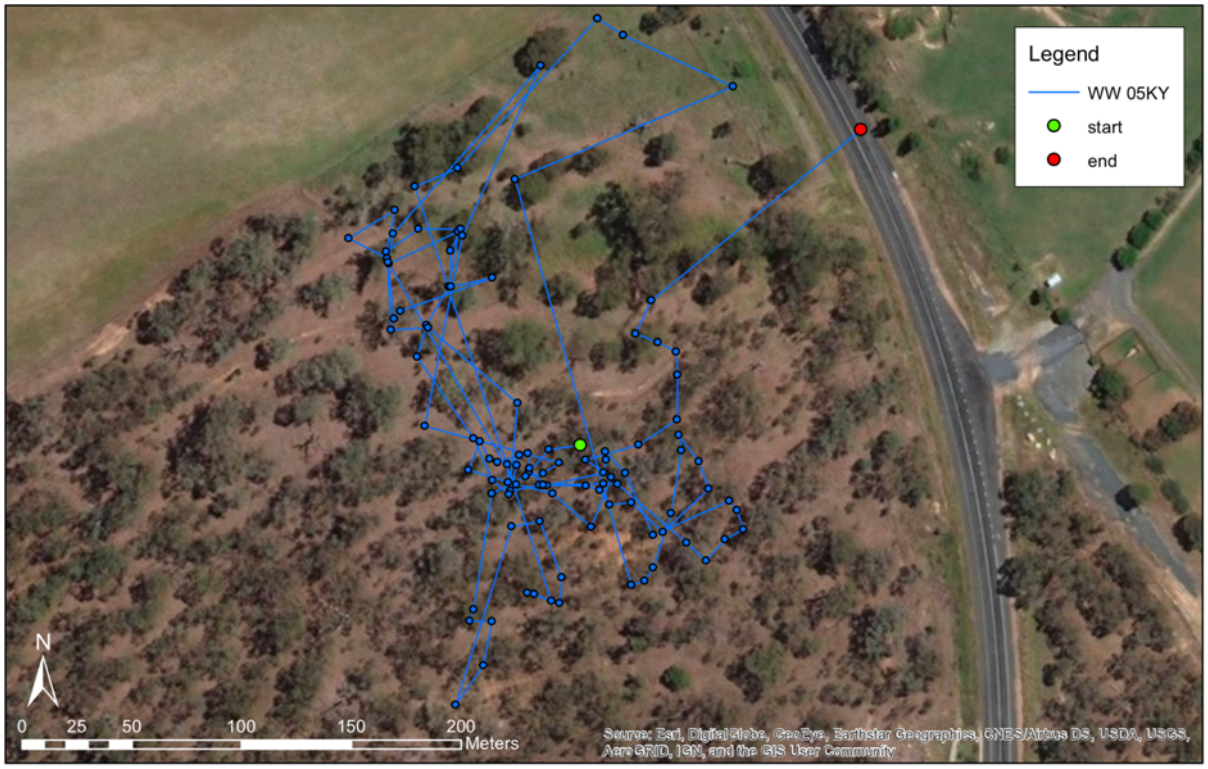
Reference sites






Legend
2016
WW 05KY
WW 04 KY
2015
WW 01KY
N
0 50 100 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community


Legend
WW 04KY
start
end
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

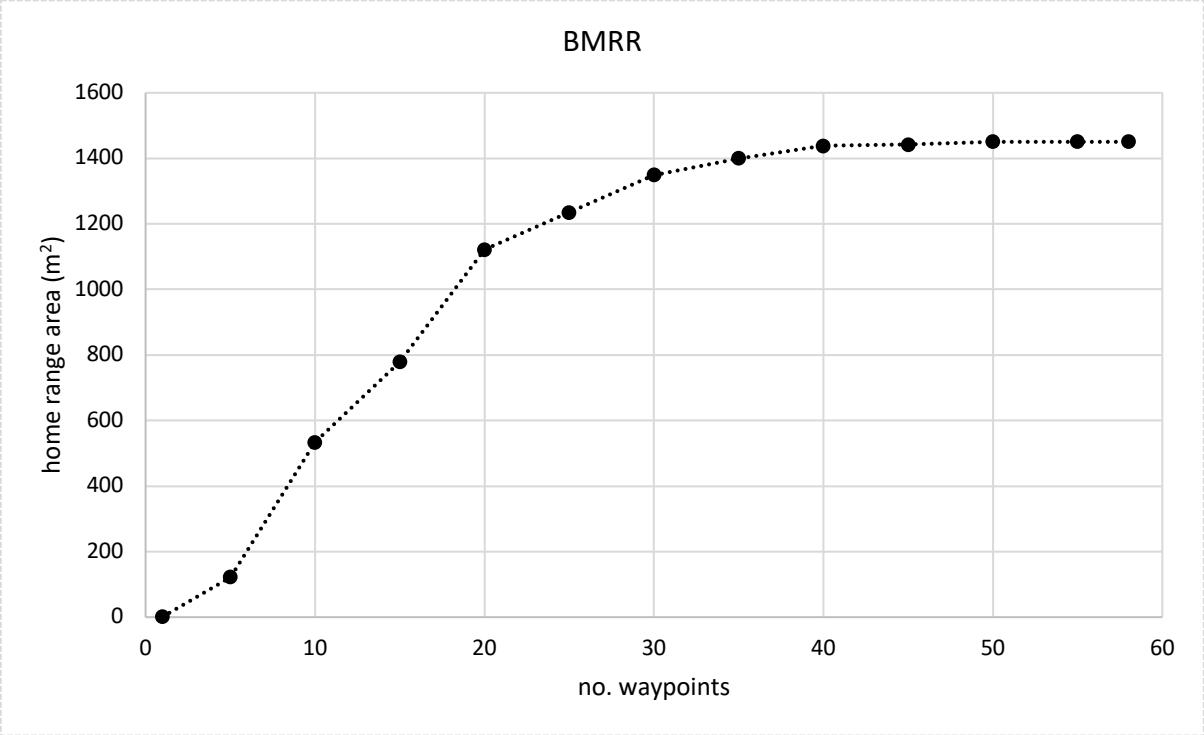

Legend
WW 05KY
start
end
N
0 25 50 100 150 200
Meters
Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

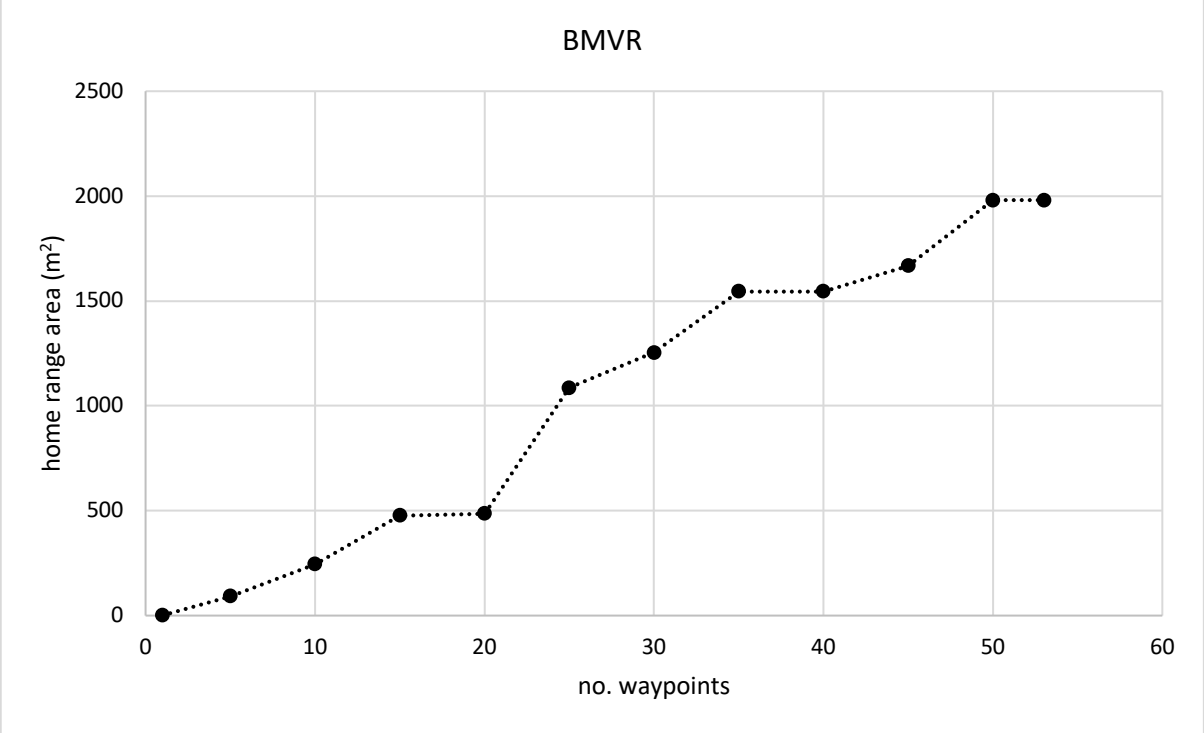
APPENDIX E

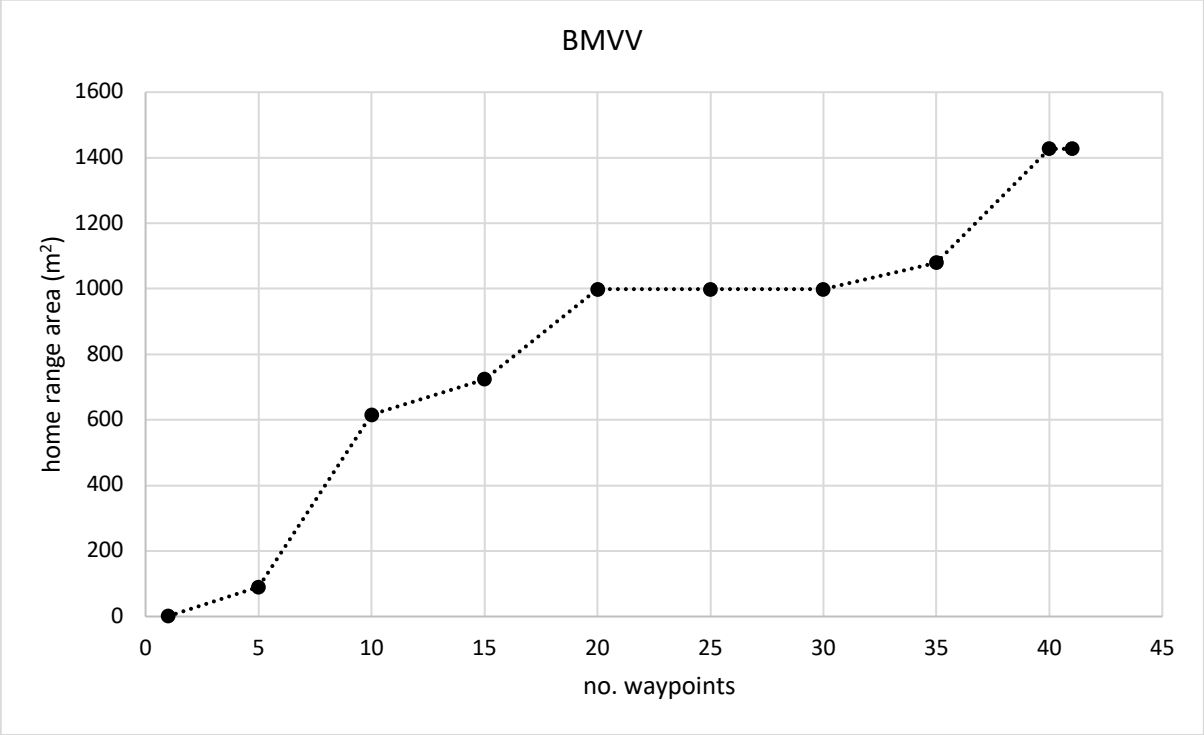
Accumulation curves obtained in 2015 pilot study of superb fairywren (*Malurus cyaneus*) and willie wagtail (*Rhipidura leucophrys*) home ranges

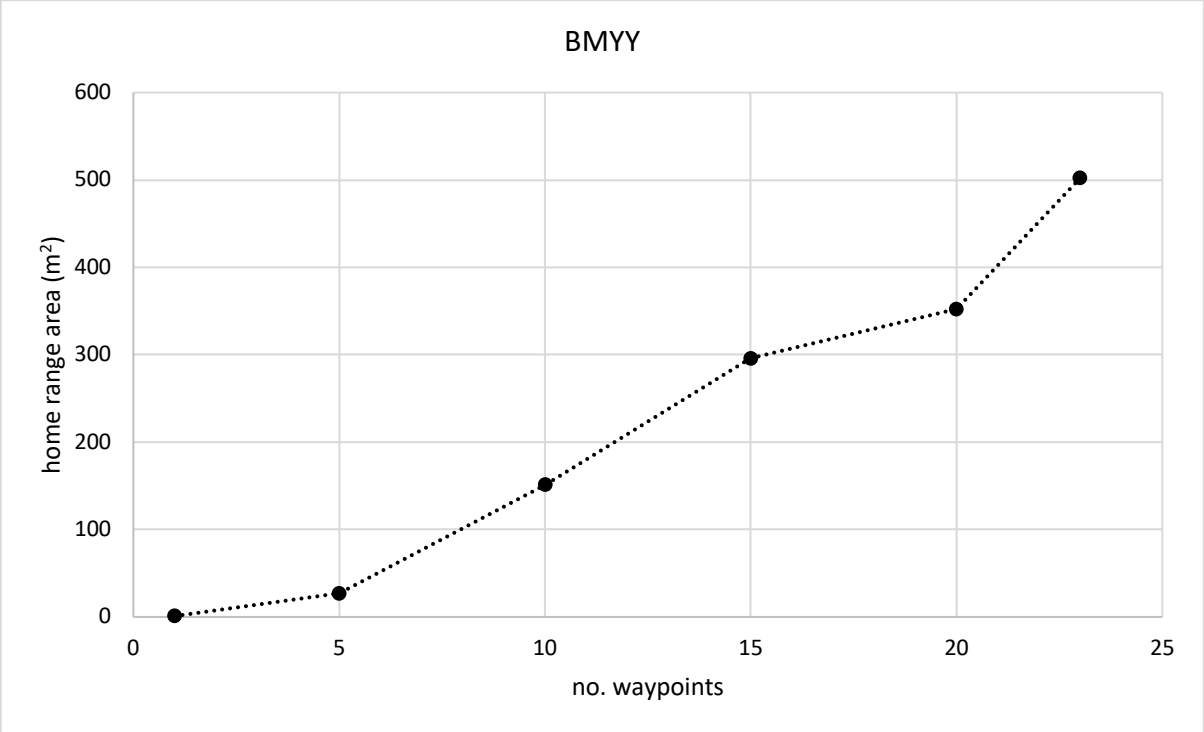
YMYY, a.k.a. “Gary”, a male superb fairywren (*Malurus cyaneus*) whose home range was tracked during this study. Photo: Donna Belder.

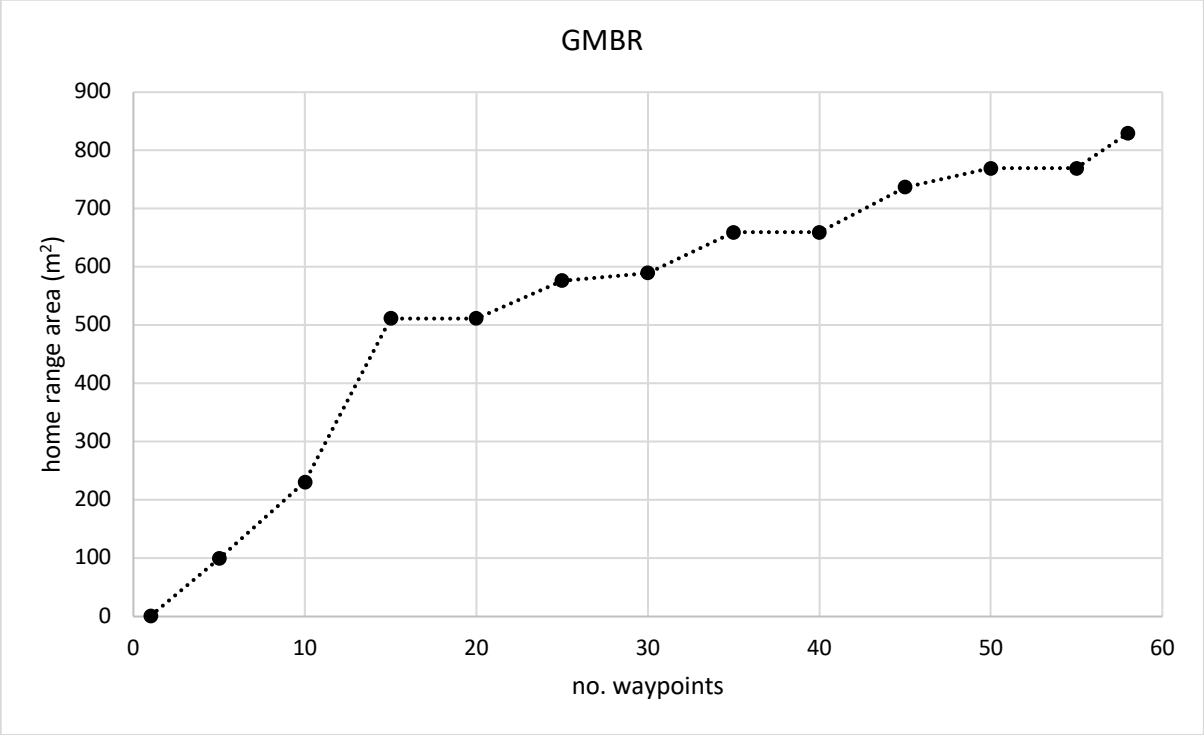
Superb fairywren home ranges

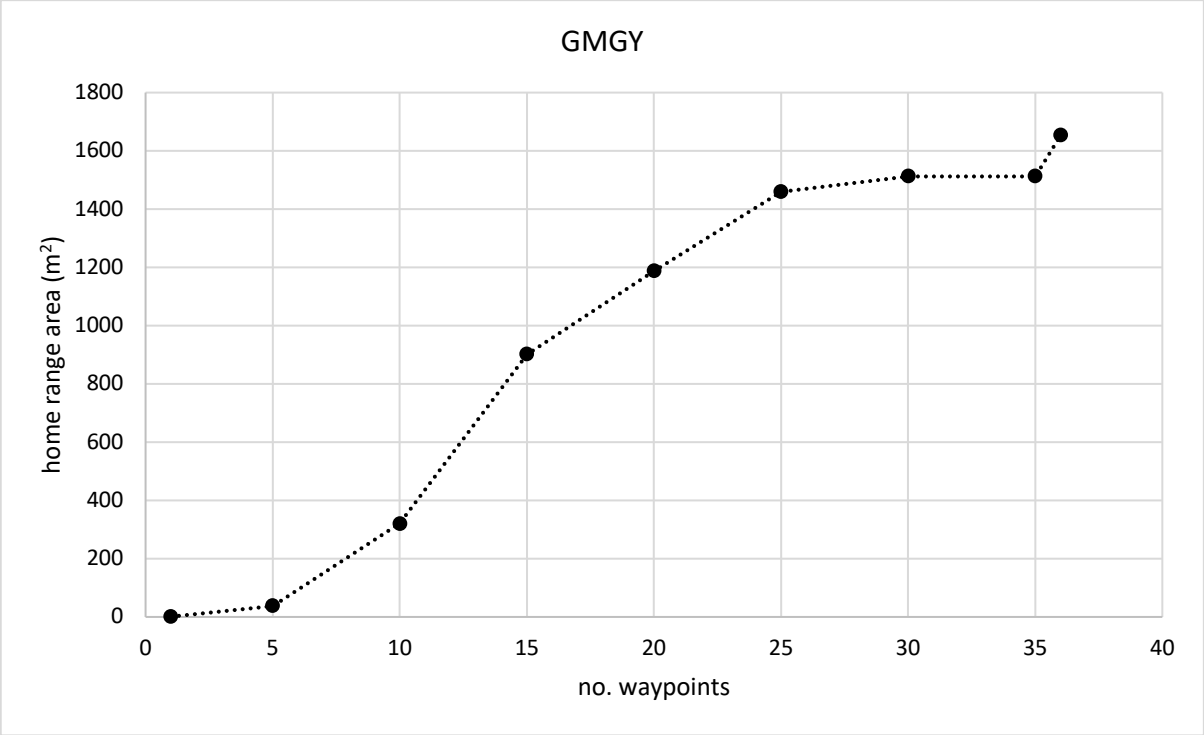


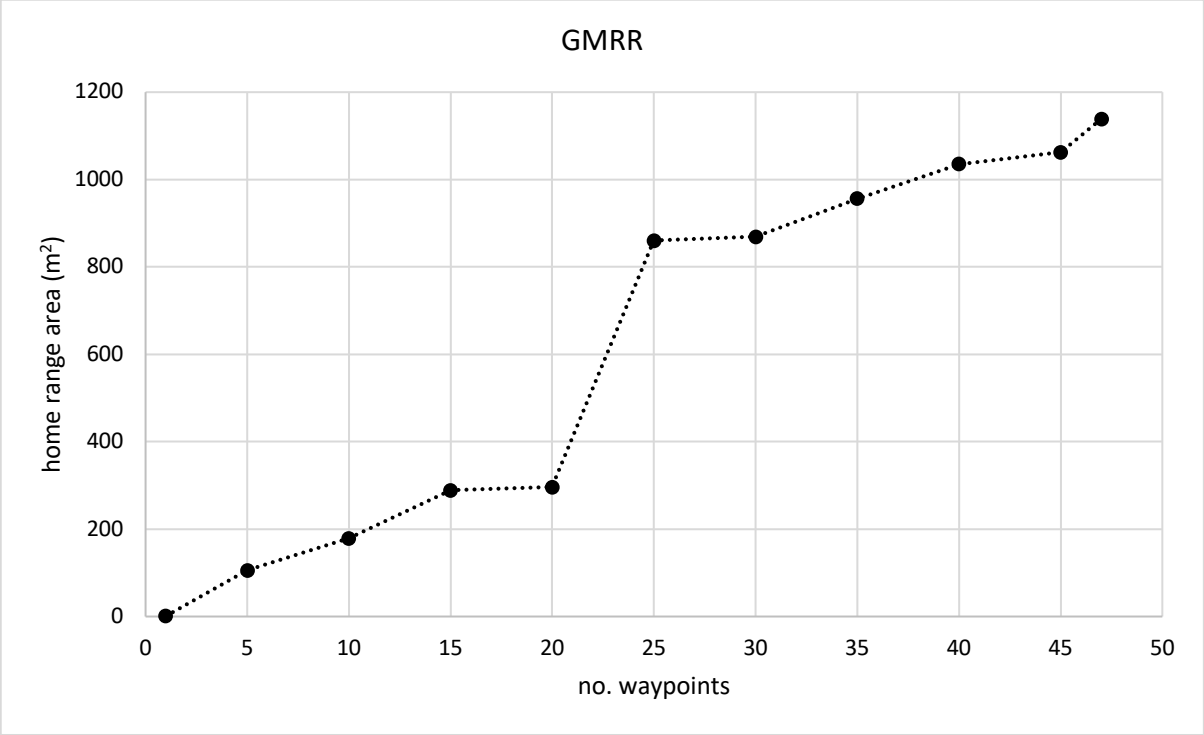


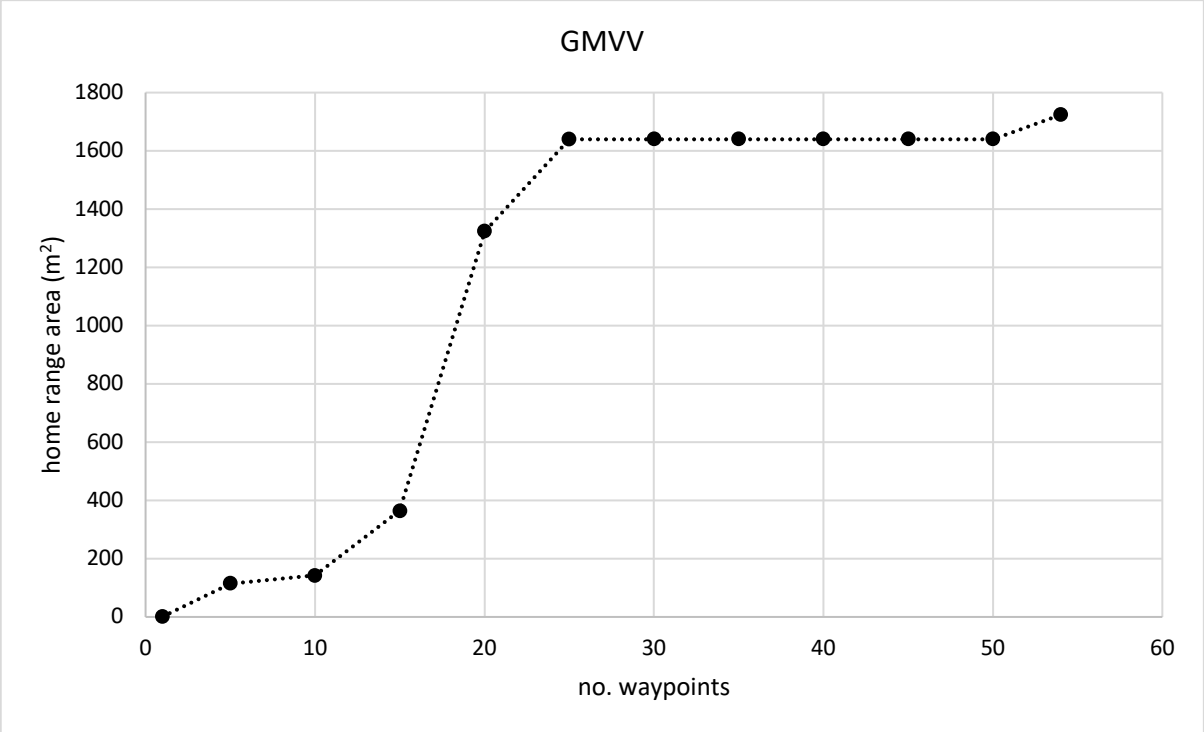

BMVV
home range area (m²)
1600
1400
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
35
40
45
no. waypoints

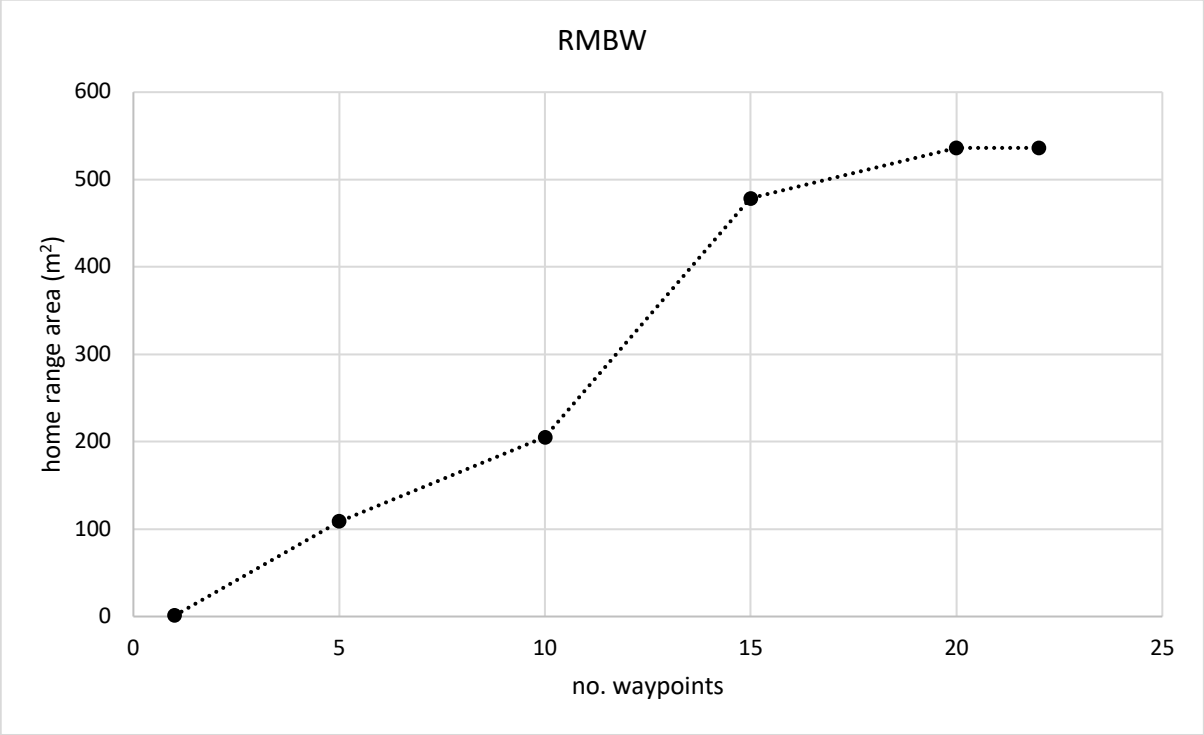

BMYY
home range area (m²)
600
500
400
300
200
100
0
0
5
10
15
20
25
no. waypoints

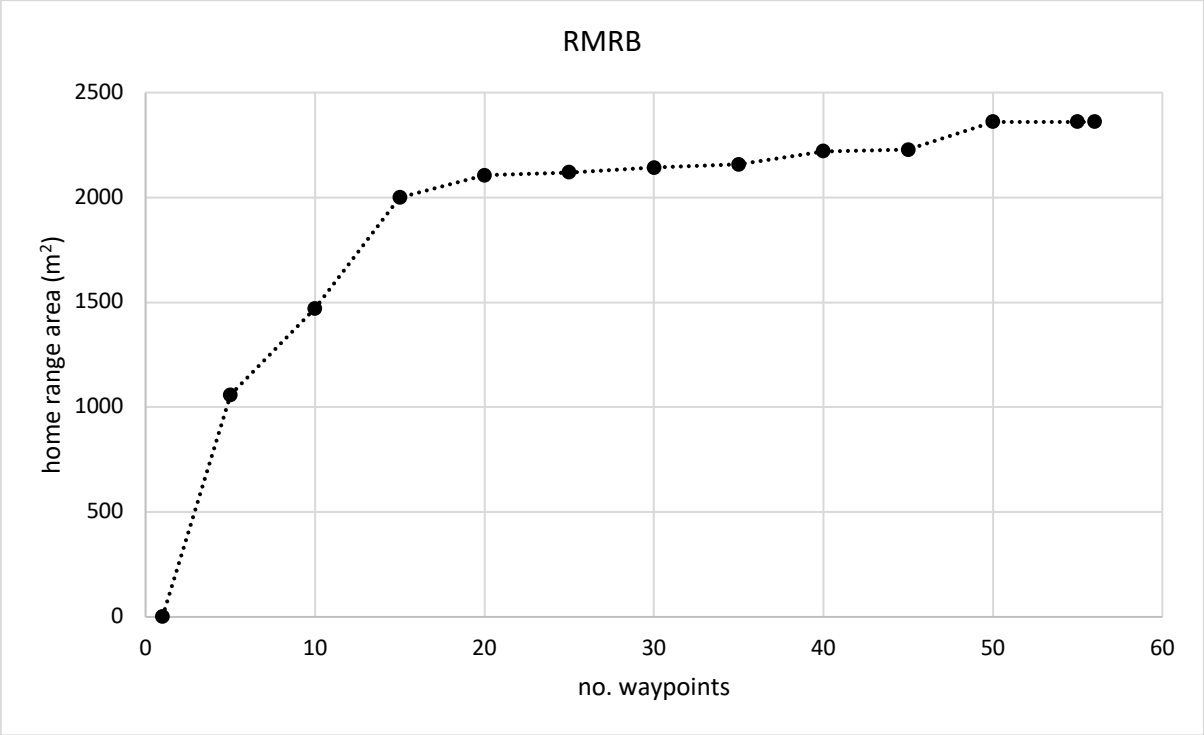

GMBR
home range area (m²)
900
800
700
600
500
400
300
200
100
0
0
10
20
30
40
50
60
no. waypoints

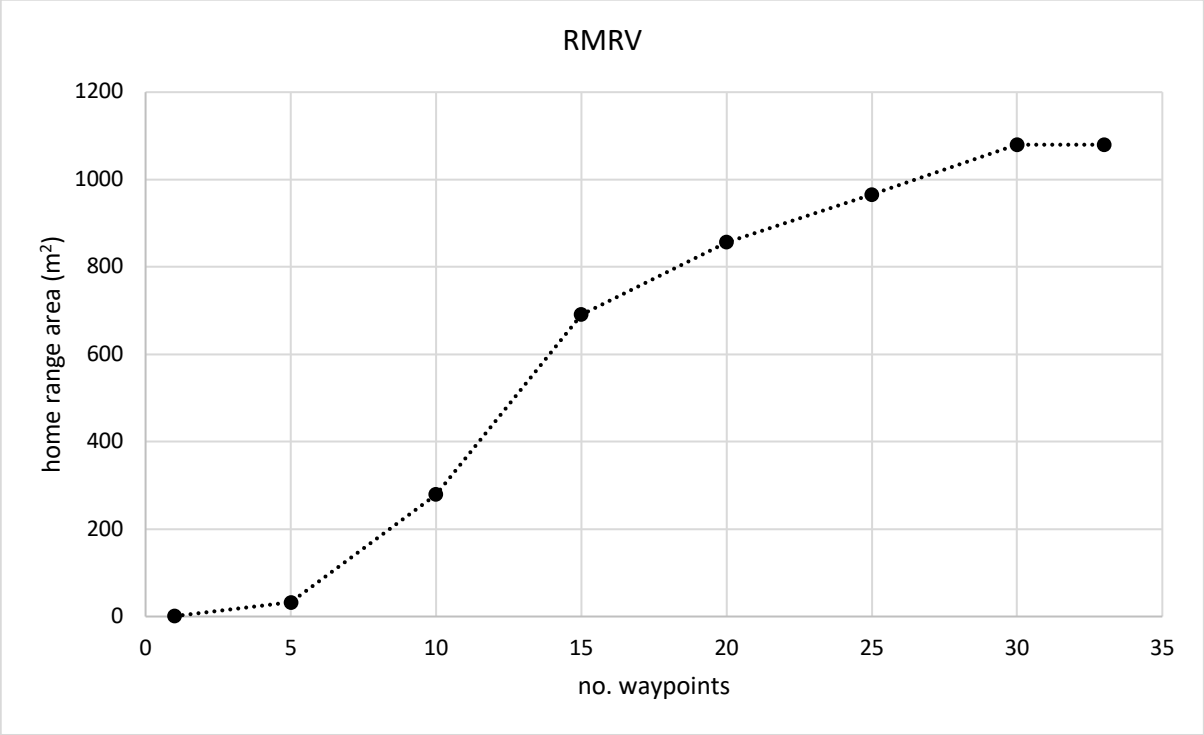

GMGY
home range area (m²)
1800
1600
1400
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
35
40
no. waypoints

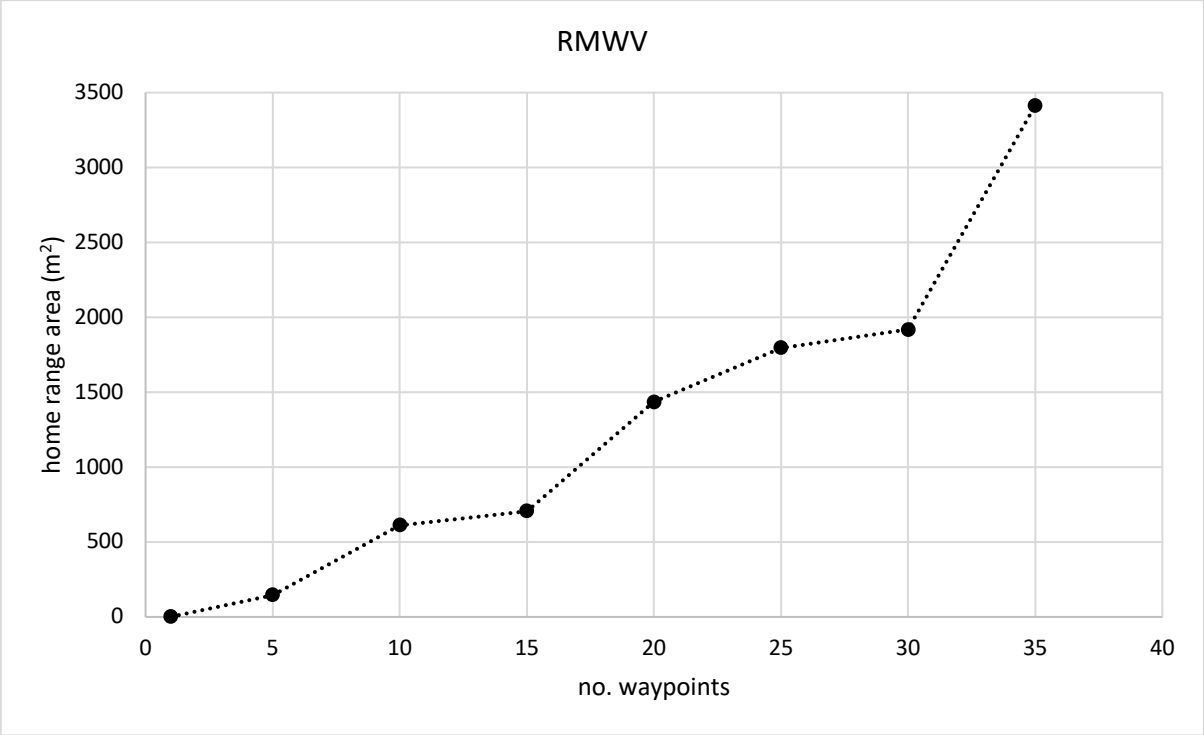

GMRR
home range area (m²)
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
35
40
45
50
no. waypoints

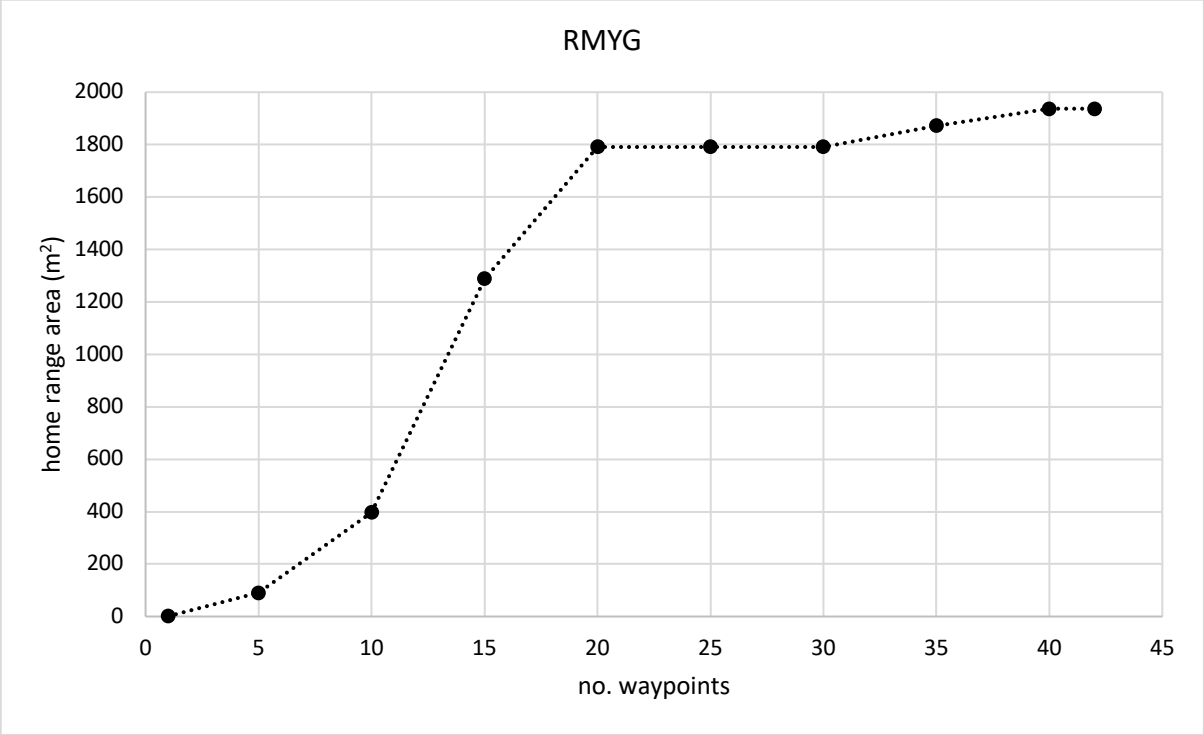

GMVV
home range area (m²)
1800
1600
1400
1200
1000
800
600
400
200
0
0
10
20
30
40
50
60
no. waypoints

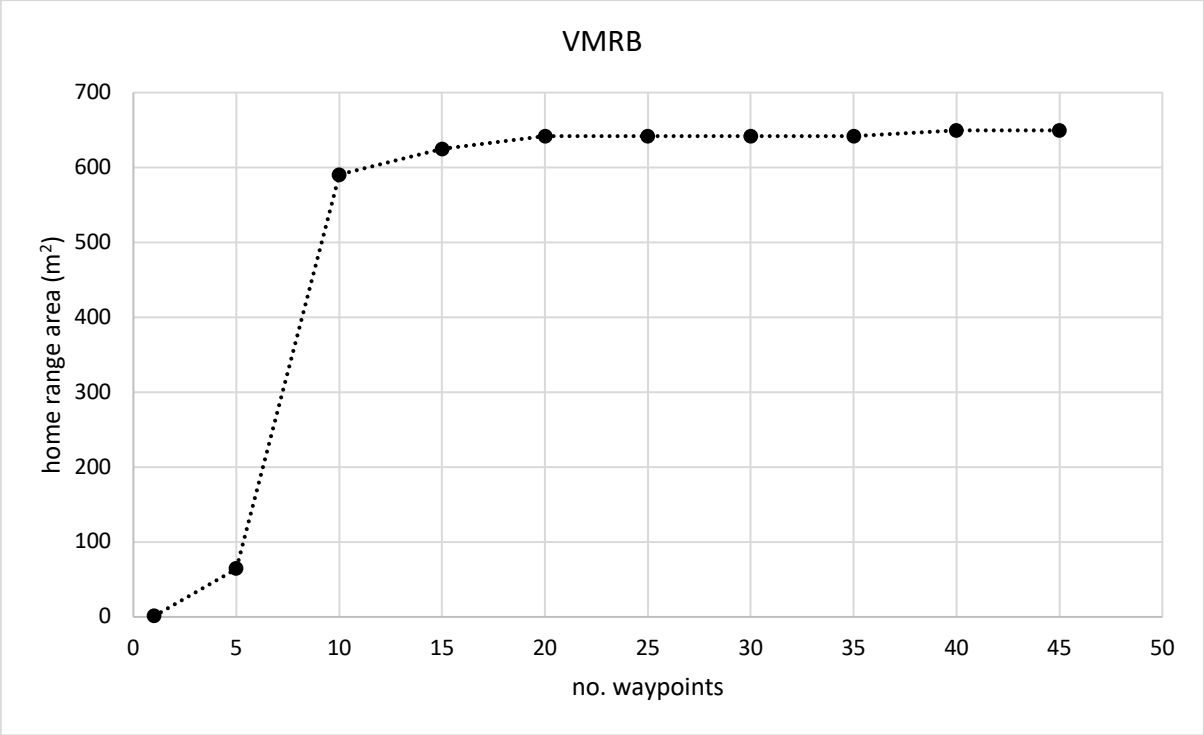

RMBW
600
500
400
300
200
100
0
home range area (m²)
0
5
10
15
20
25
no. waypoints

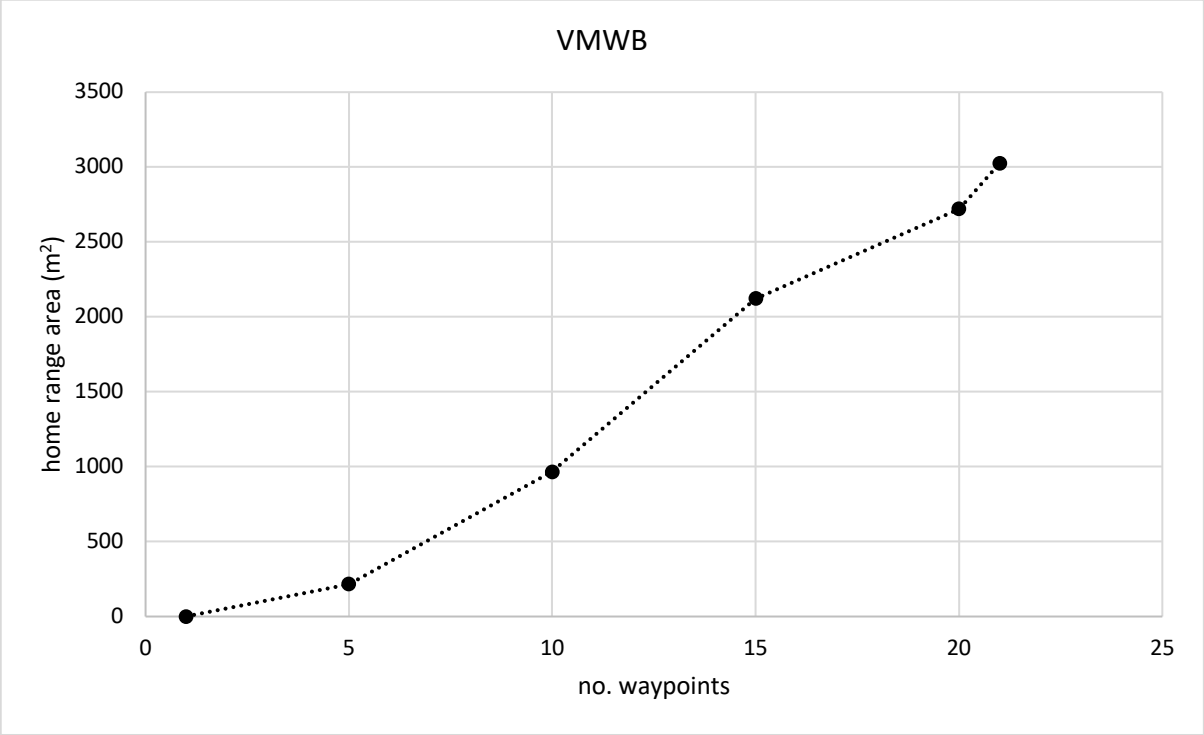

RMRB
2500
2000
1500
1000
500
0
home range area (m²)
0
10
20
30
40
50
60
no. waypoints

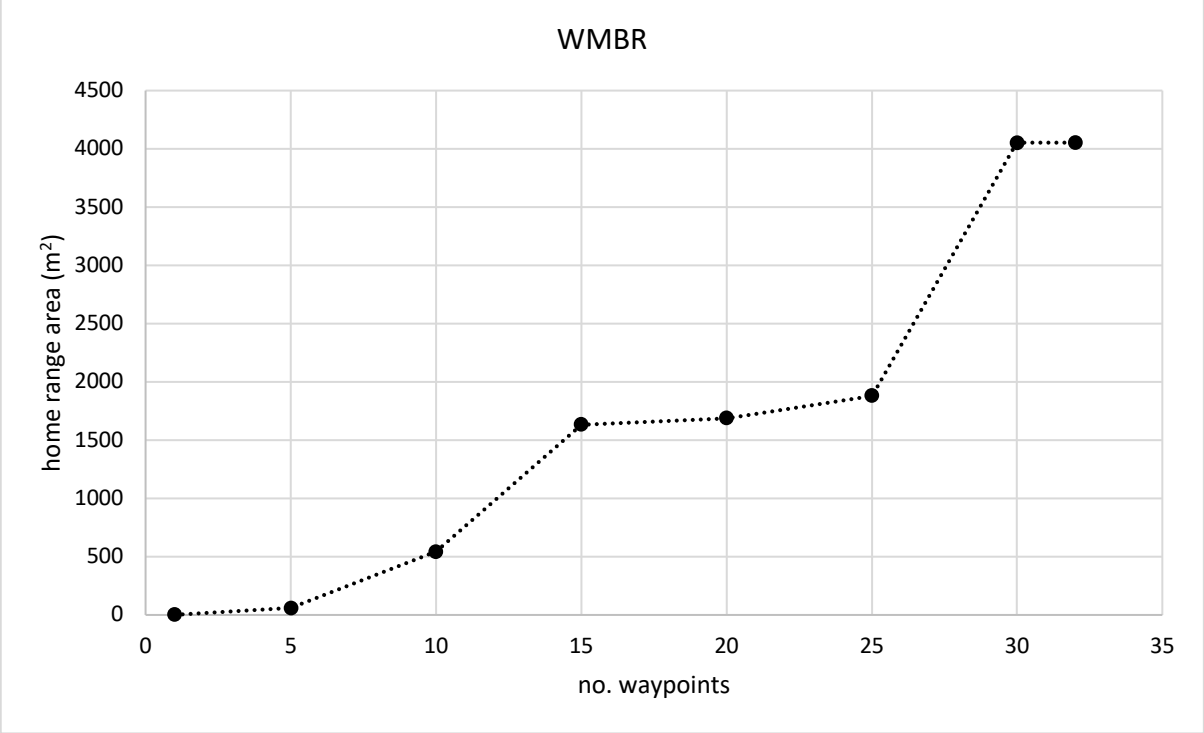

RMRV
home range area (m²)
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
35
no. waypoints

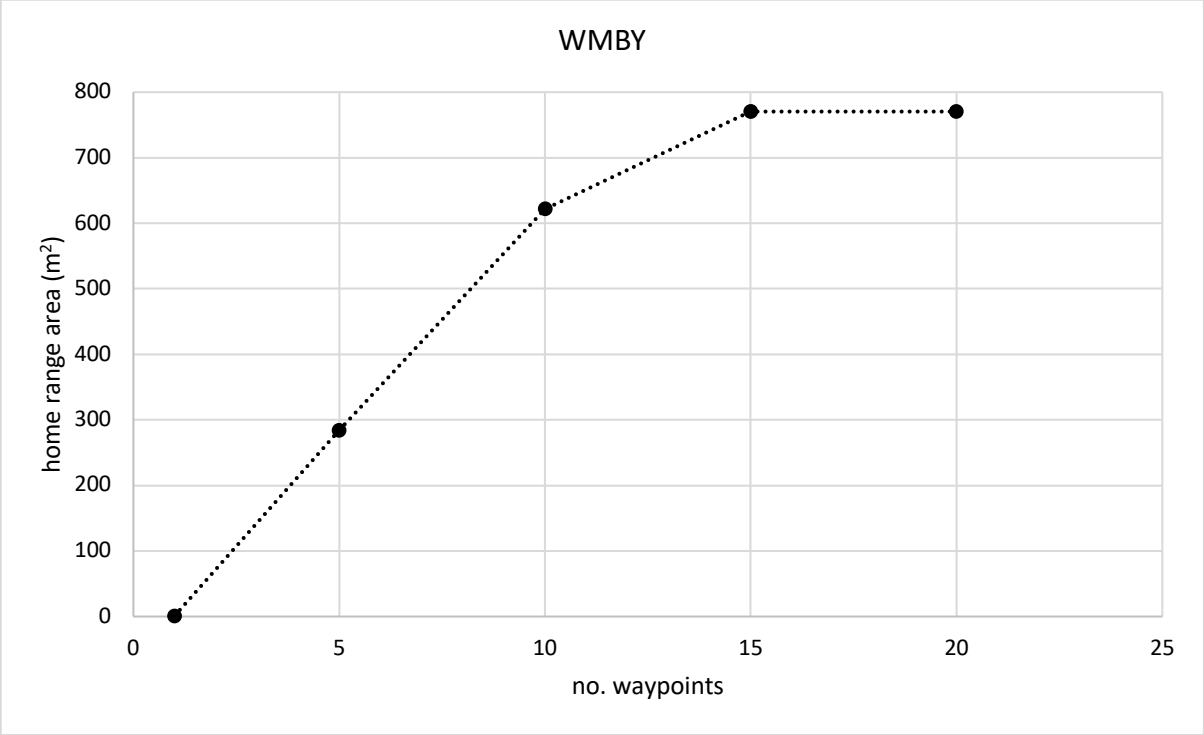

RMWV
home range area (m²)
3500
3000
2500
2000
1500
1000
500
0
0
5
10
15
20
25
30
35
40
no. waypoints

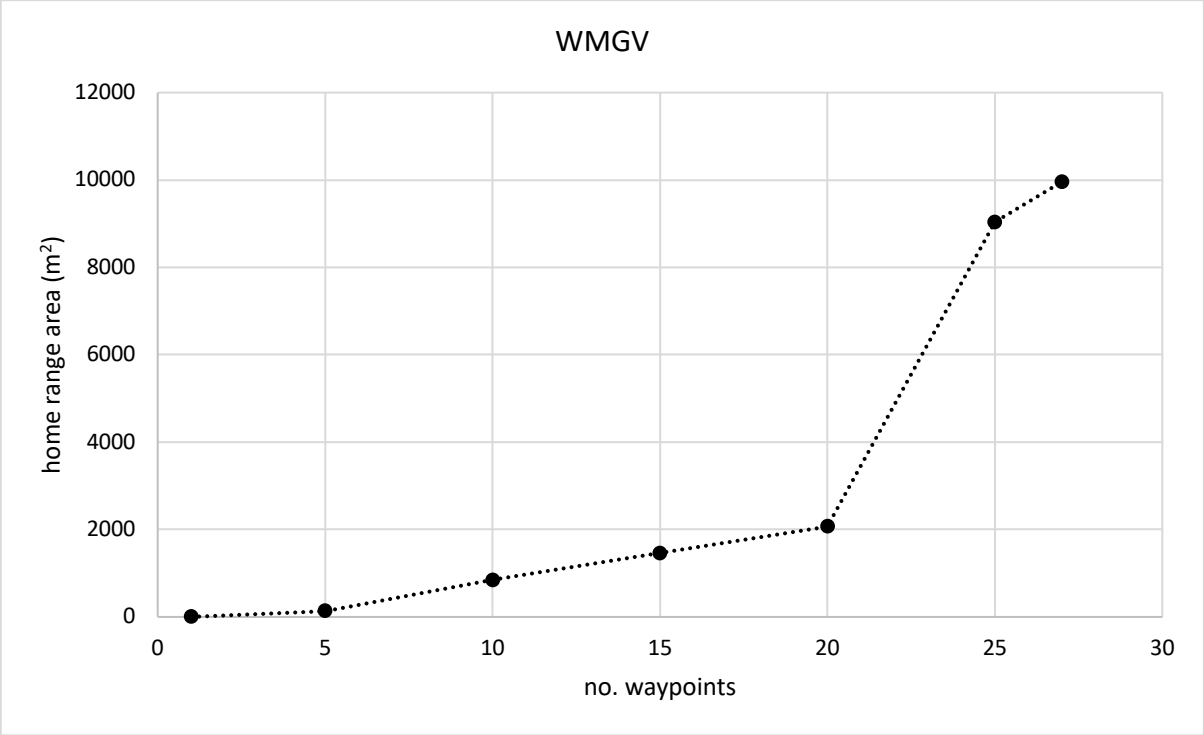

RMYG
home range area (m²)
2000
1800
1600
1400
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
35
40
45
no. waypoints

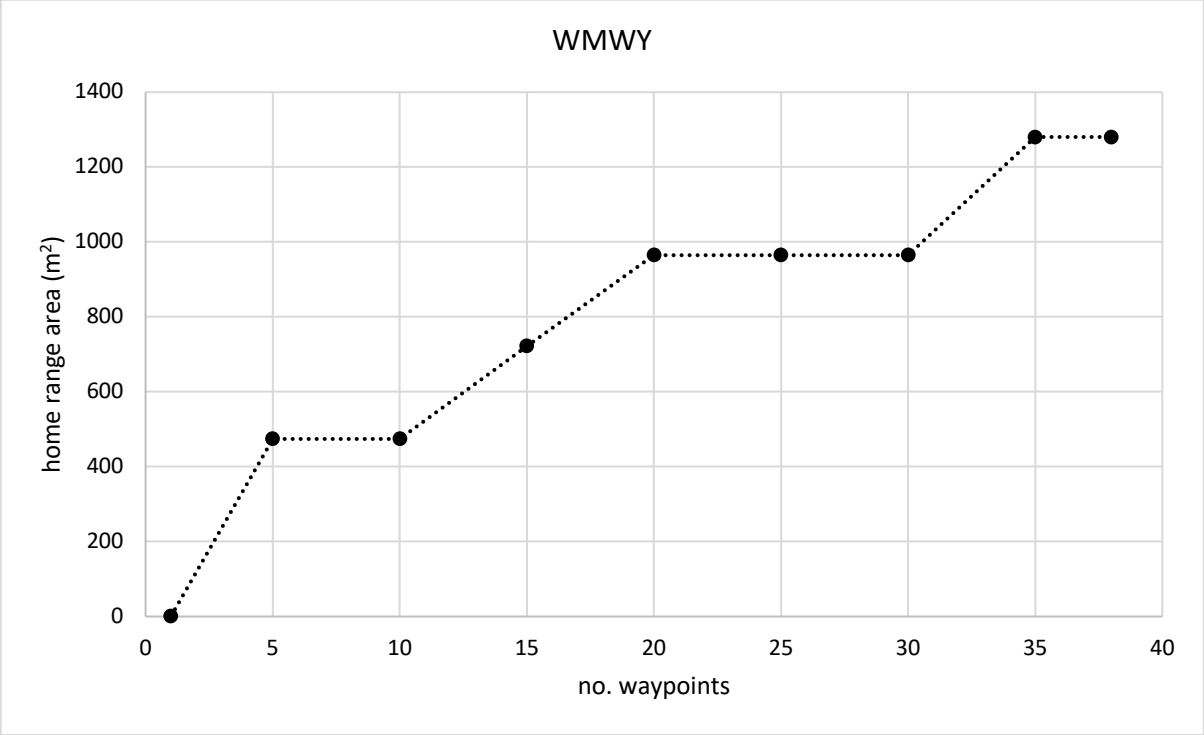

VMRB
home range area (m²)
700
600
500
400
300
200
100
0
0
5
10
15
20
25
30
35
40
45
50
no. waypoints

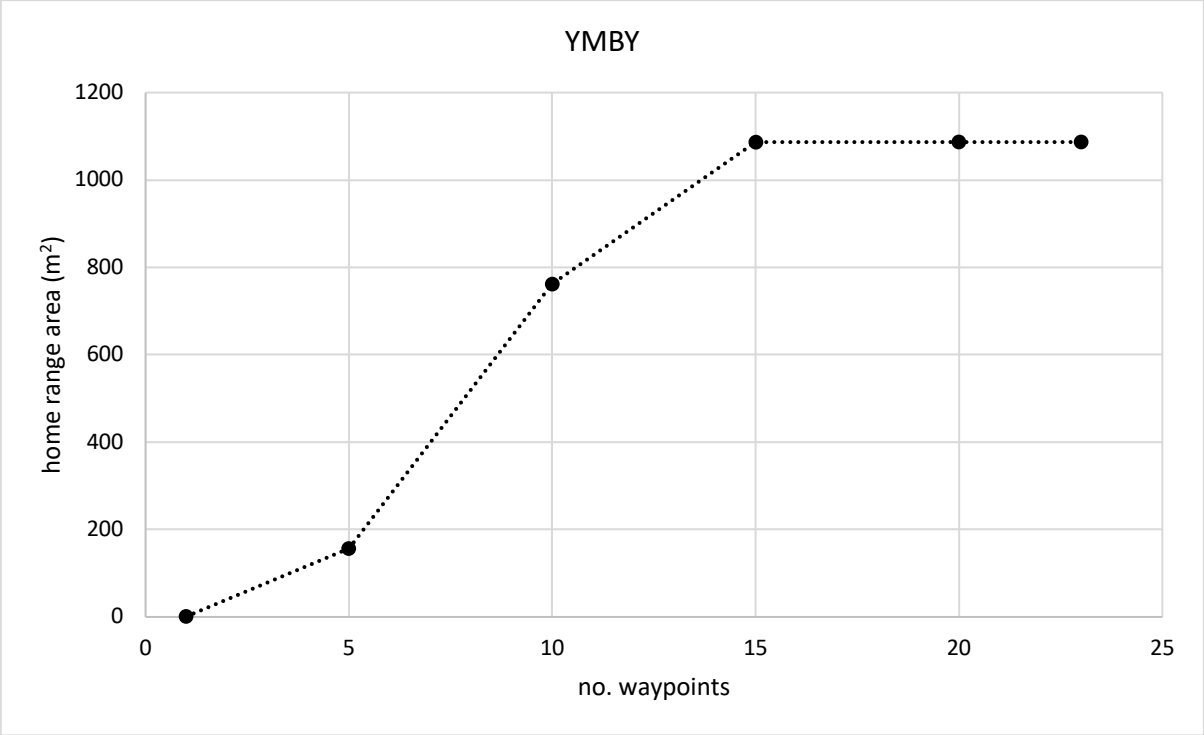

VMWB
home range area (m²)
3500
3000
2500
2000
1500
1000
500
0
0
5
10
15
20
25
no. waypoints

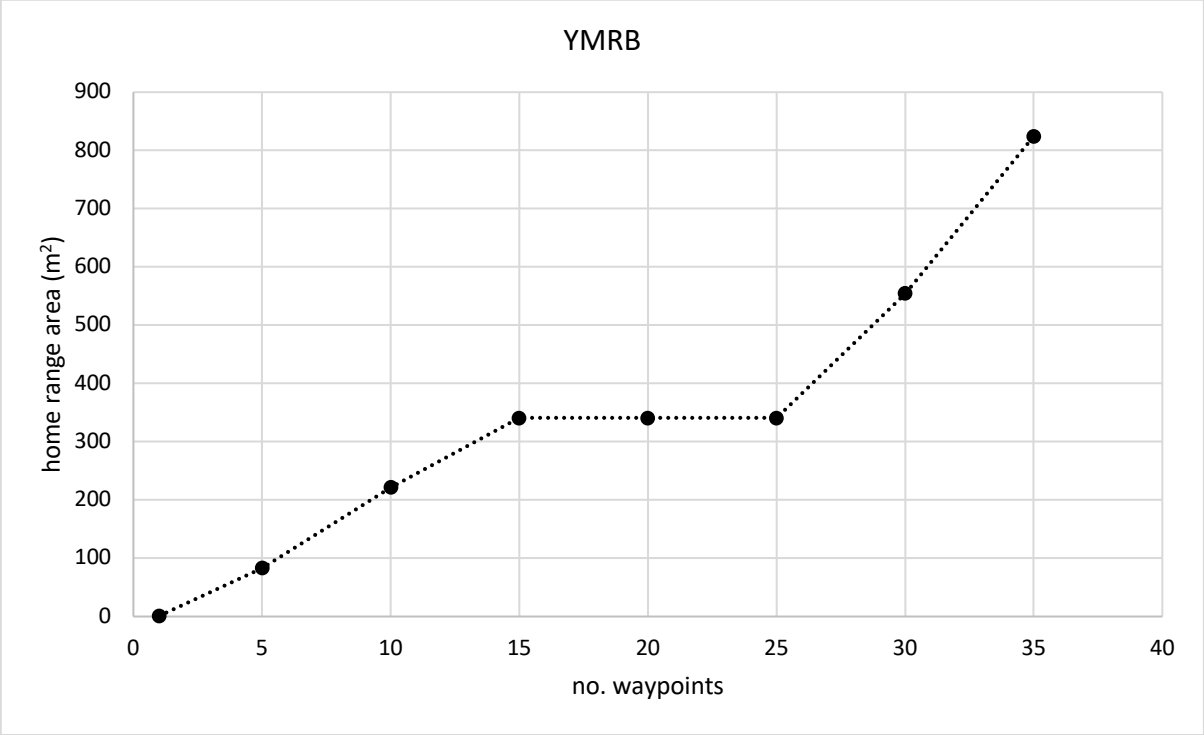

WMBR
home range area (m²)
4500
4000
3500
3000
2500
2000
1500
1000
500
0
0
5
10
15
20
25
30
35
no. waypoints

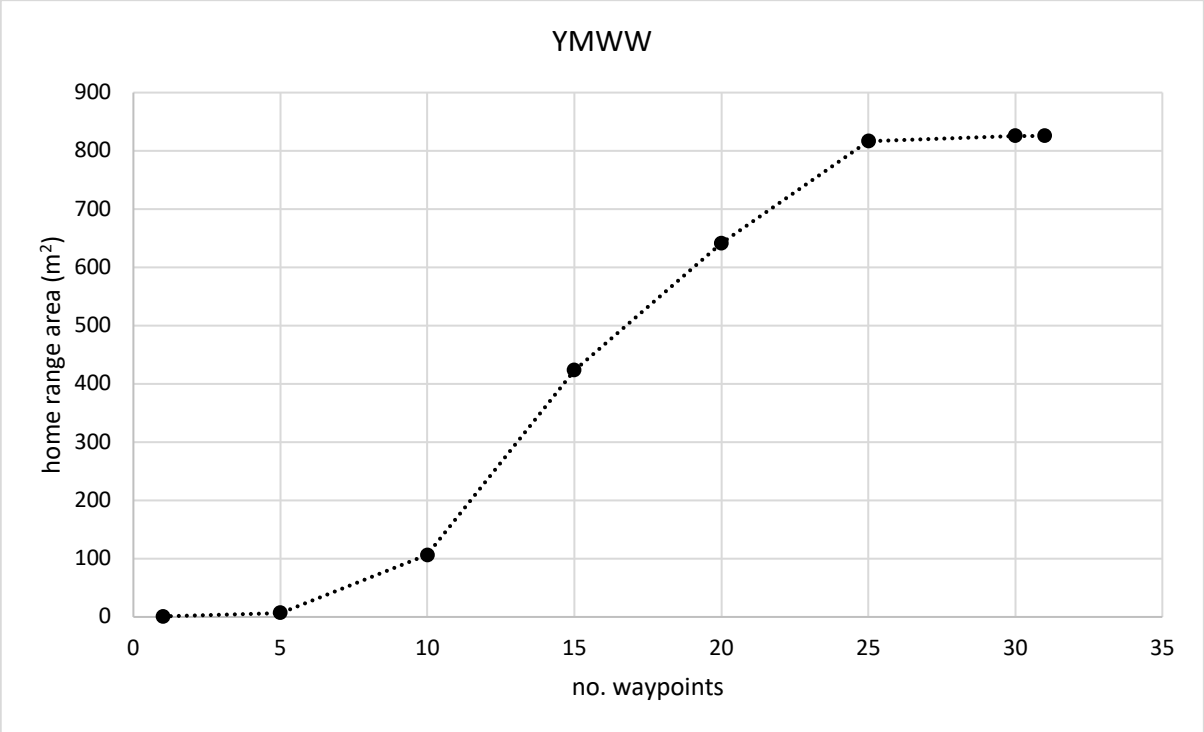

WMBY
home range area (m²)
800
700
600
500
400
300
200
100
0
0
5
10
15
20
25
no. waypoints

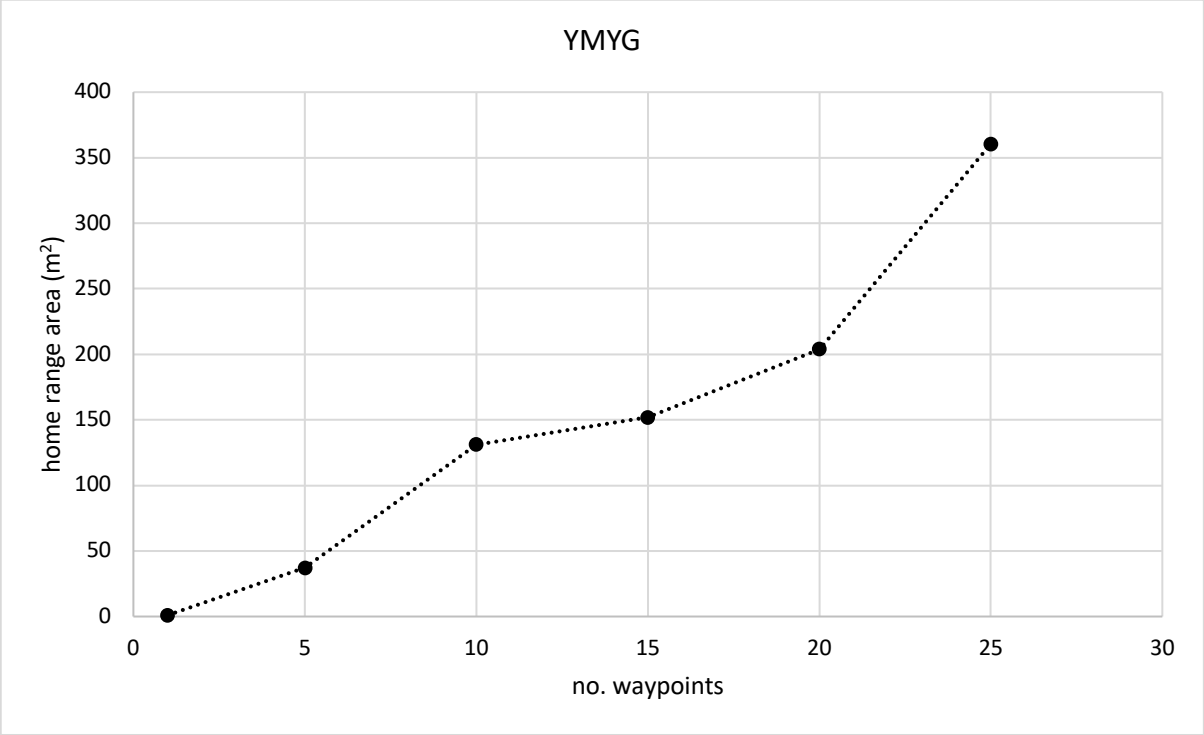

WMGV
home range area (m²)
12000
10000
8000
6000
4000
2000
0
0
5
10
15
20
25
30
no. waypoints

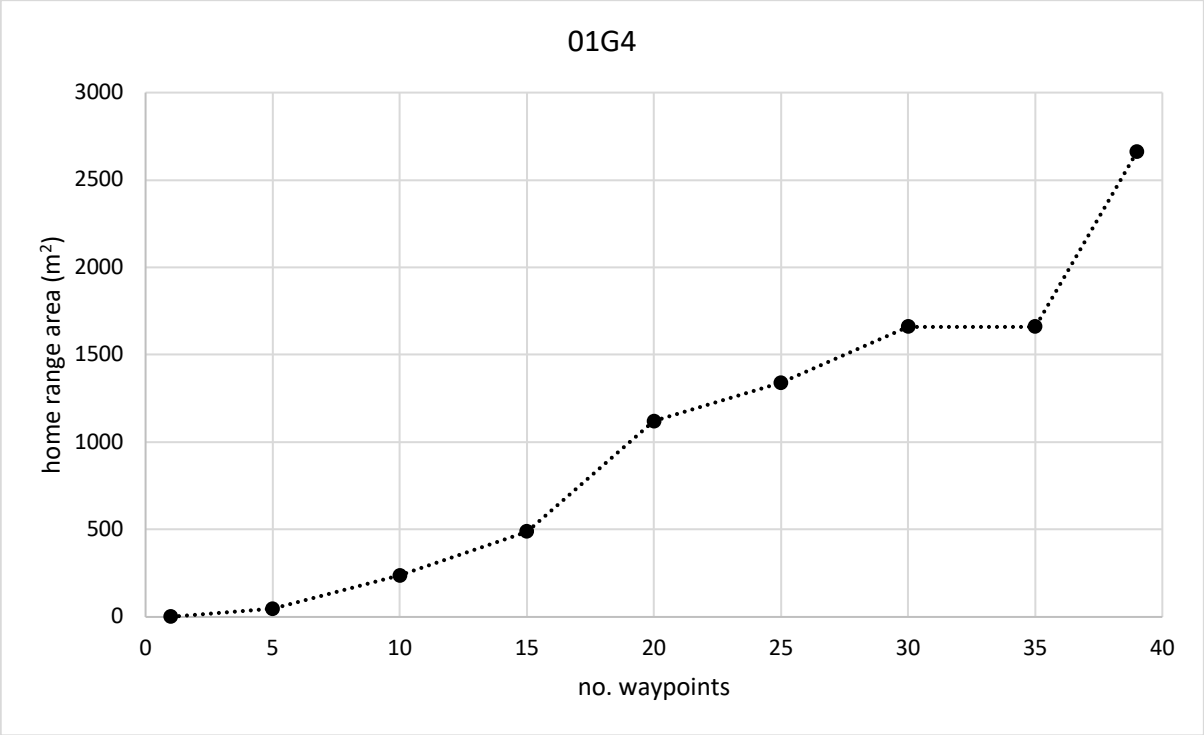

WMWY
home range area (m²)
1400
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
35
40
no. waypoints

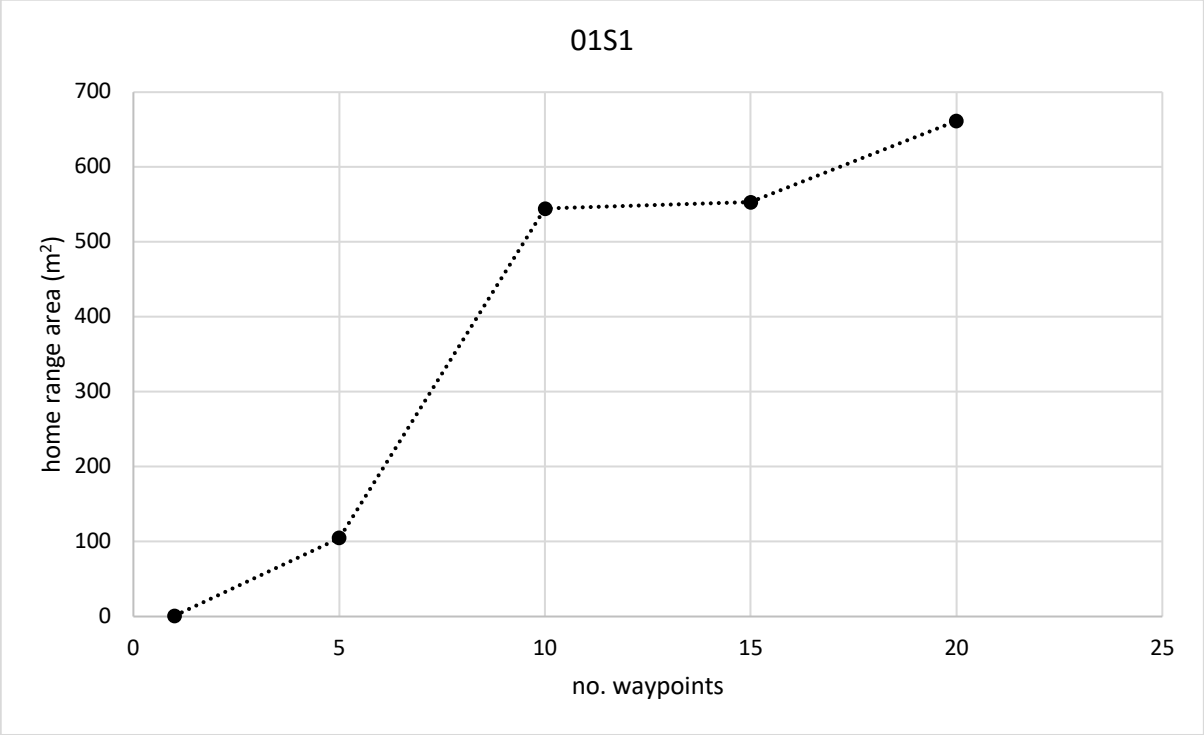

YMBY
home range area (m²)
1200
1000
800
600
400
200
0
0
5
10
15
20
25
no. waypoints


YMRB
home range area (m²)
900
800
700
600
500
400
300
200
100
0
0
5
10
15
20
25
30
35
40
no. waypoints

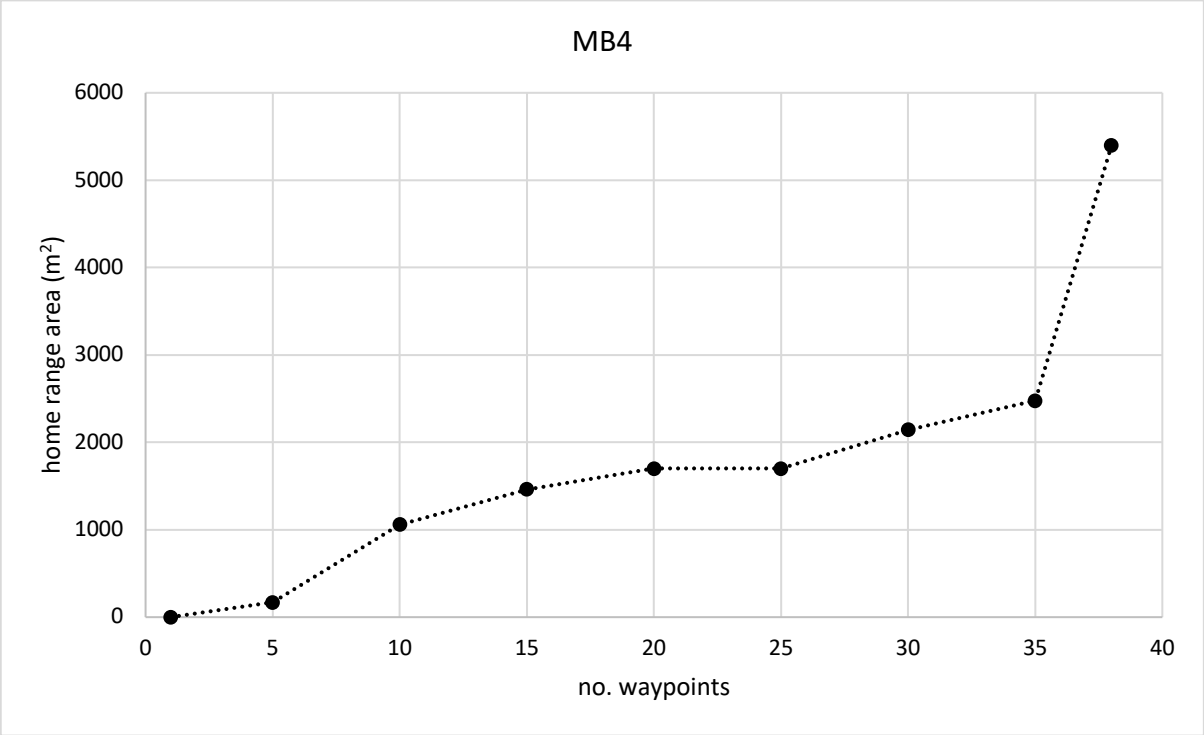

YMWW
home range area (m²)
900
800
700
600
500
400
300
200
100
0
0
5
10
15
20
25
30
35
no. waypoints

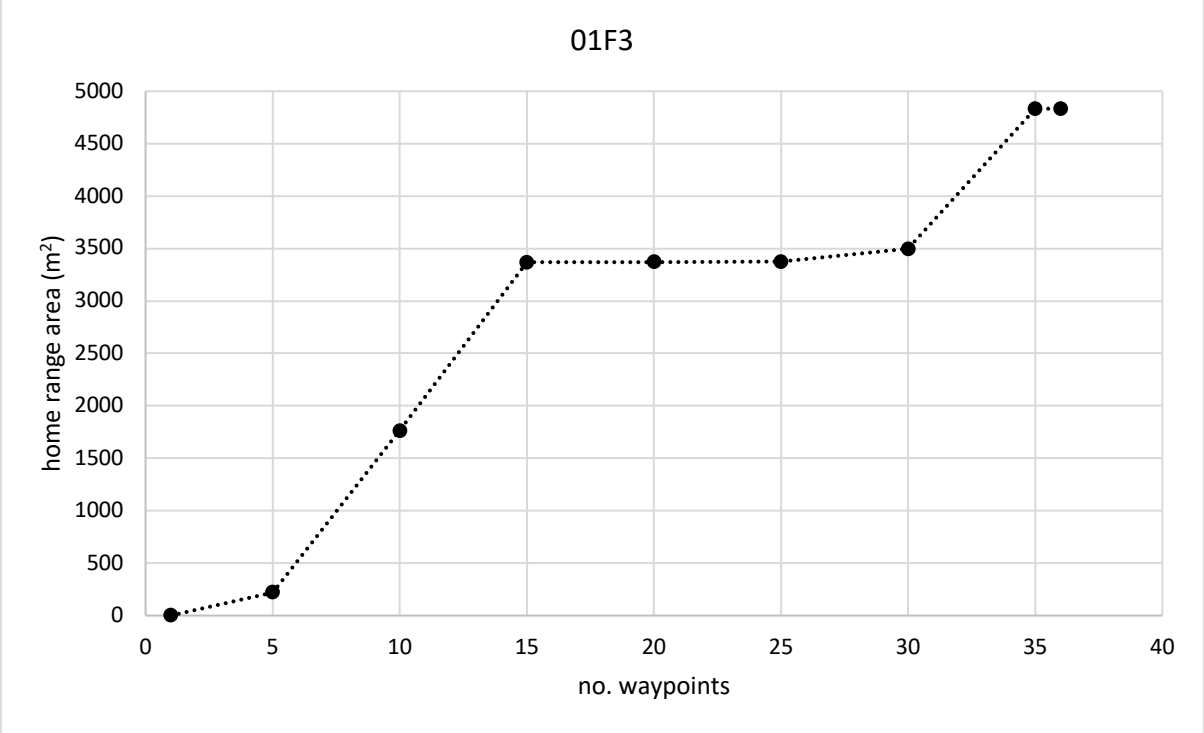

YMYG
home range area (m²)
400
350
300
250
200
150
100
50
0
0
5
10
15
20
25
30
no. waypoints

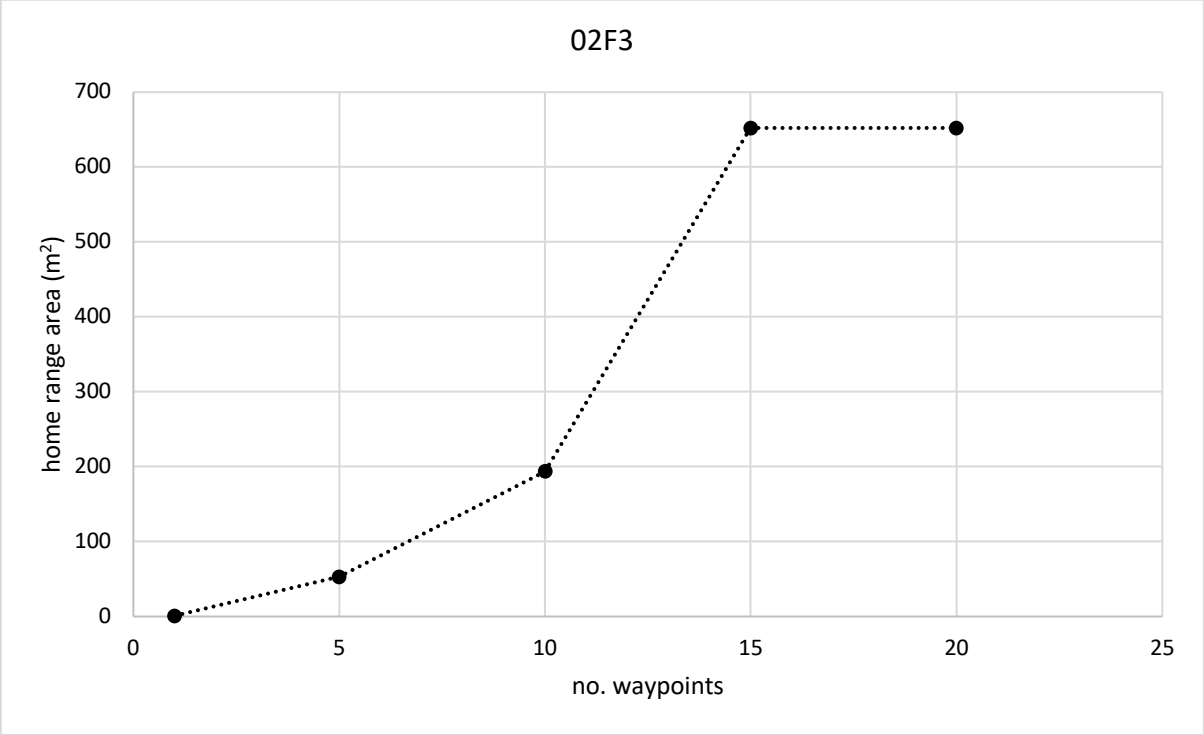

01G4
home range area (m²)
3000
2500
2000
1500
1000
500
0
0
5
10
15
20
25
30
35
40
no. waypoints

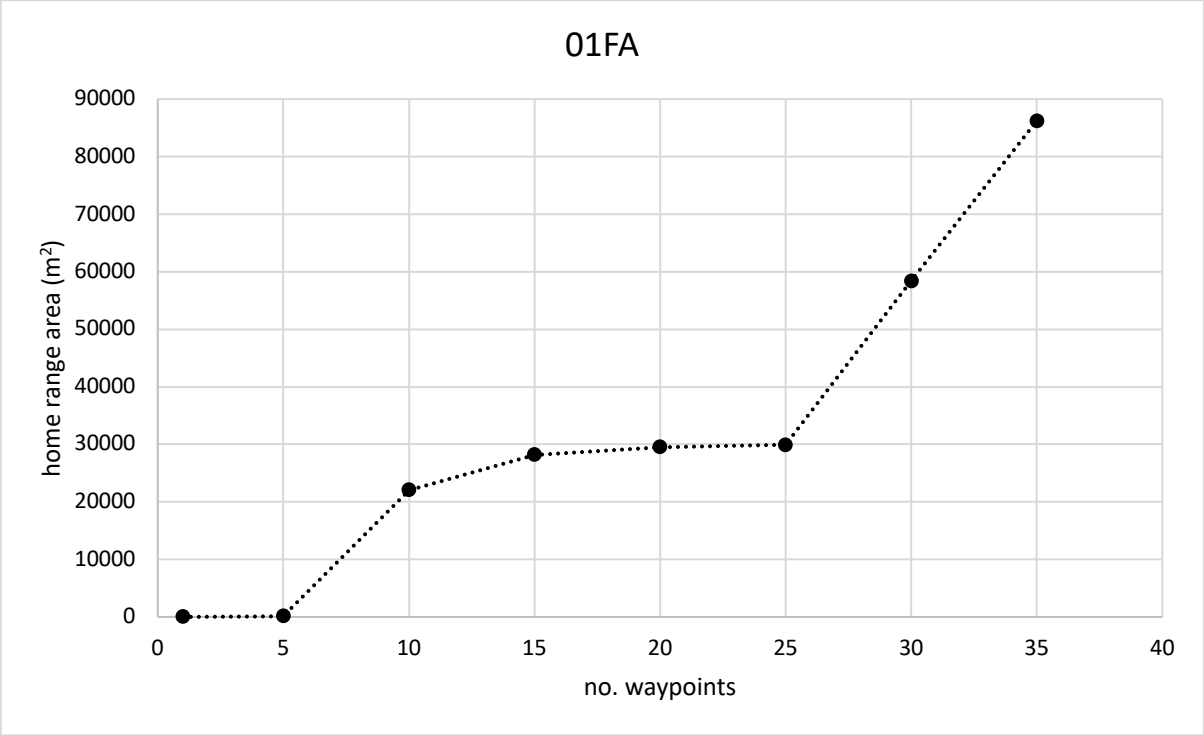

01S1
700
600
500
400
300
200
100
0
home range area (m2)
0
5
10
15
20
25
no. waypoints

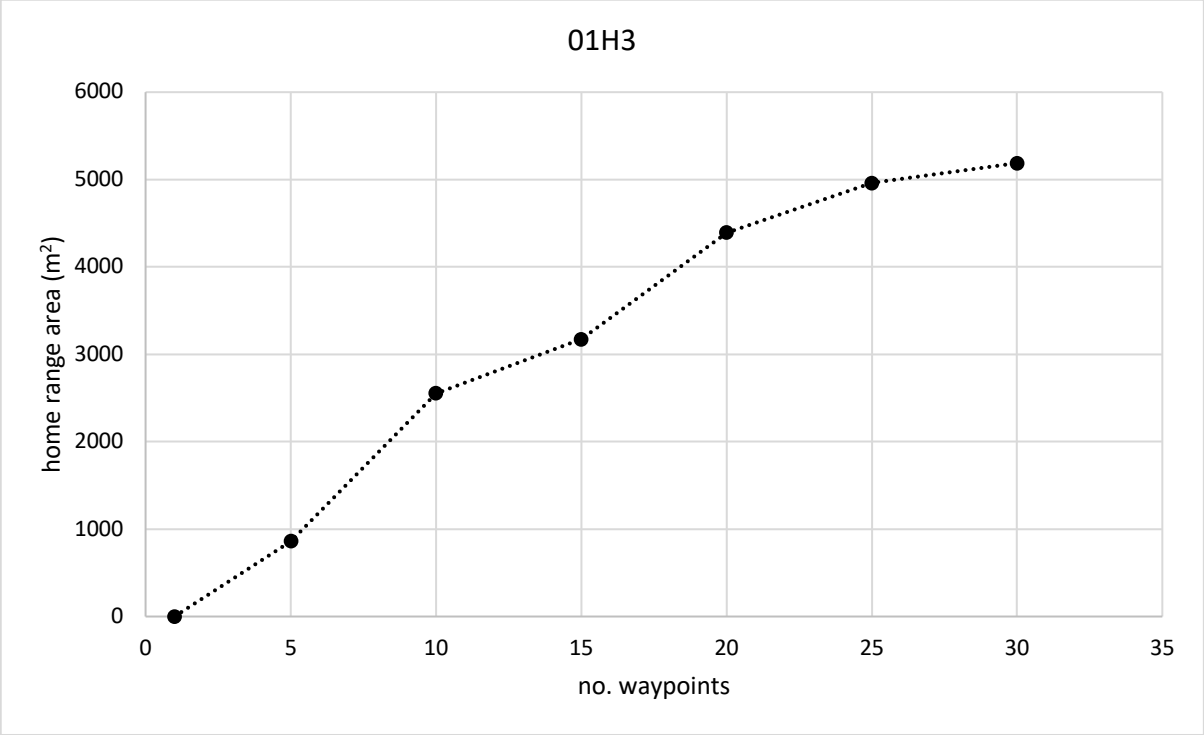
Willie wagtail home ranges

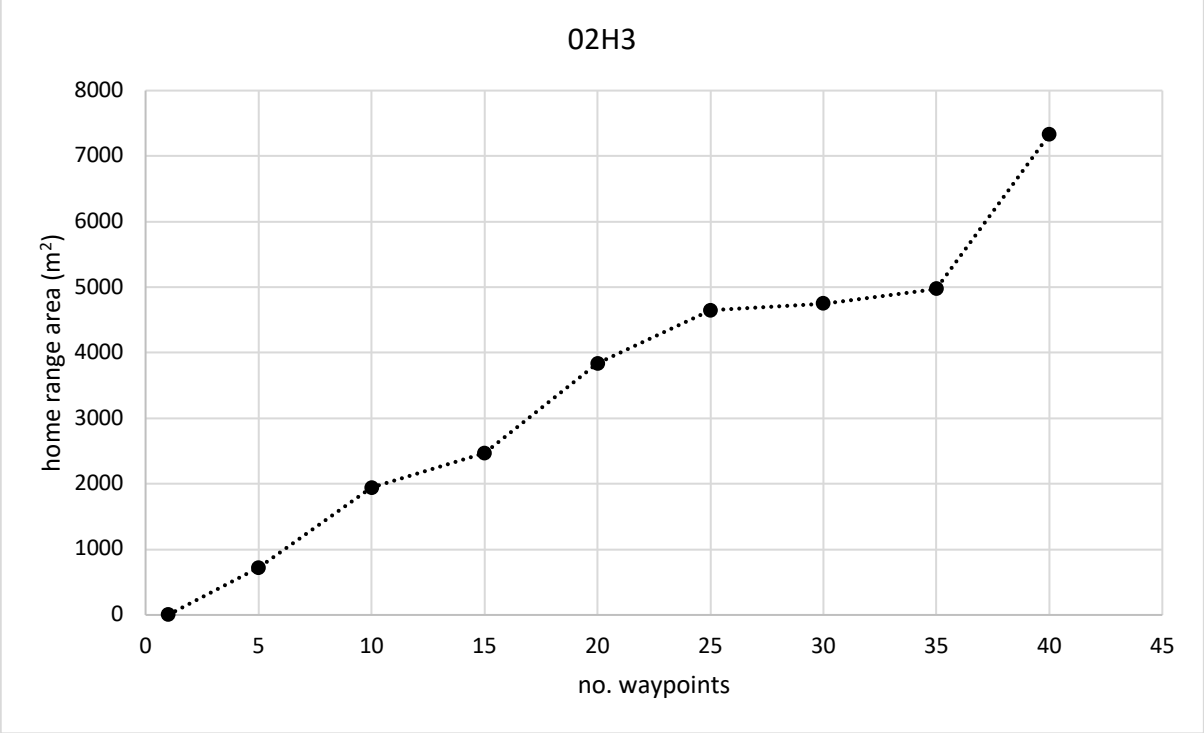


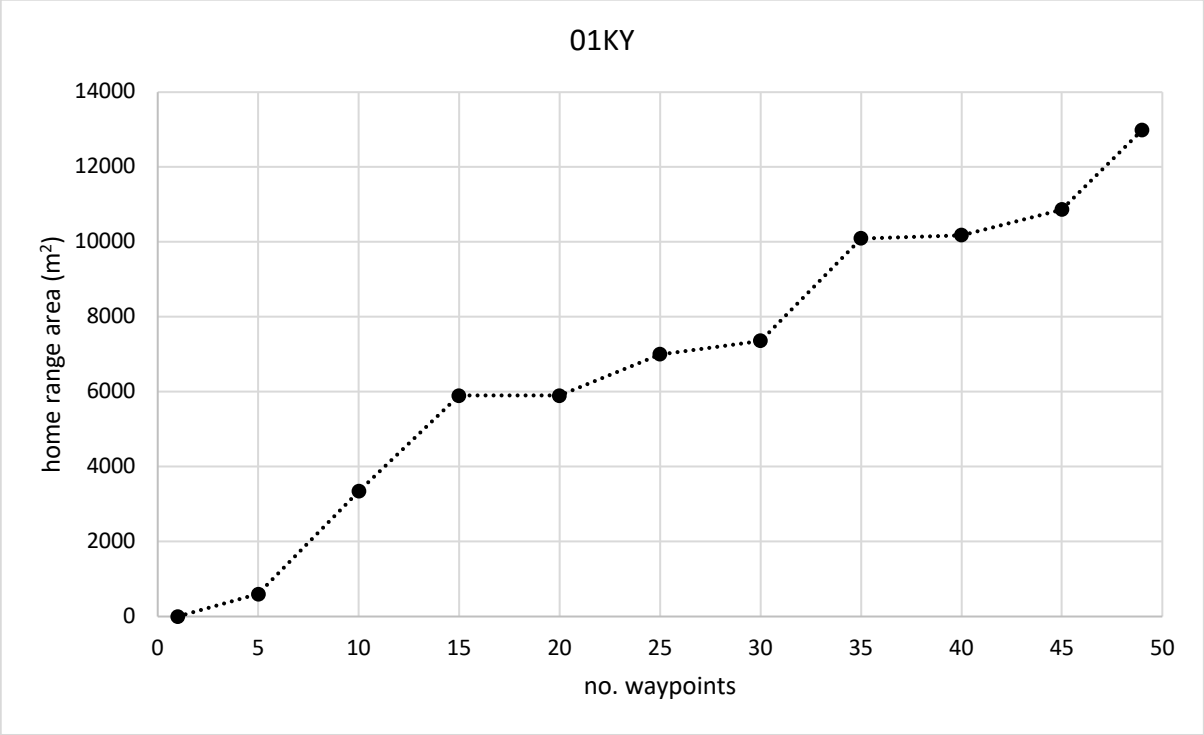


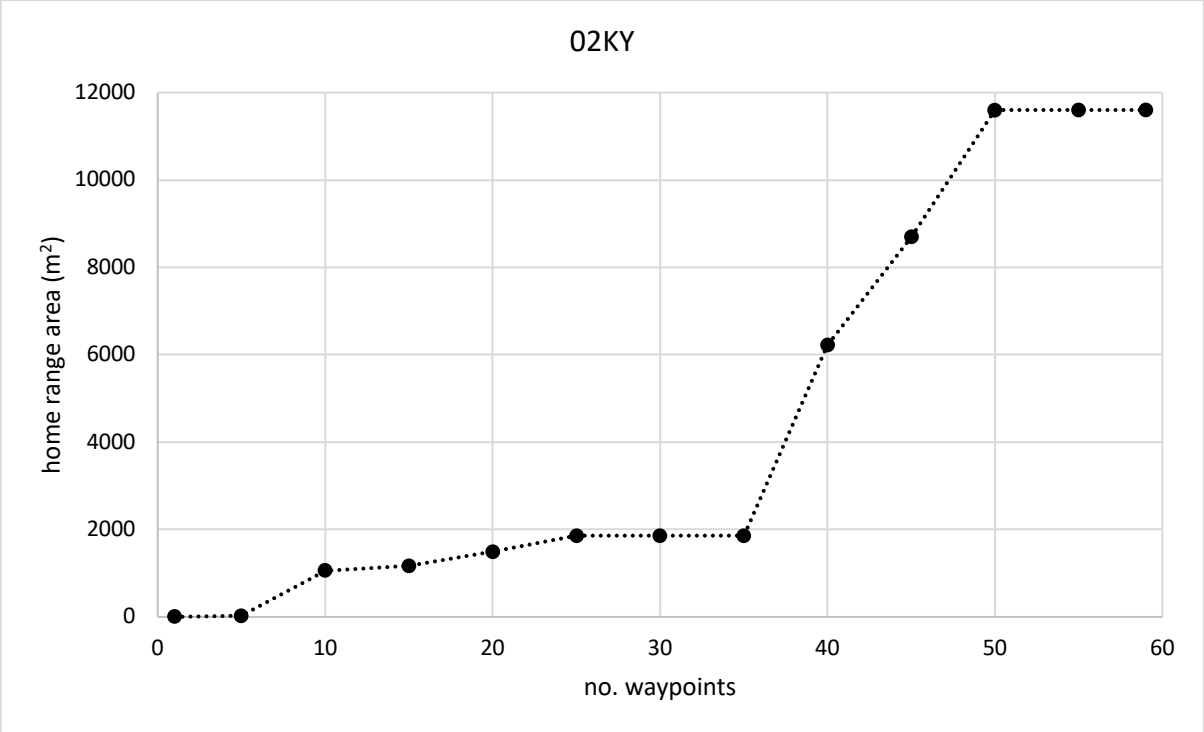

02F3
home range area (m2)
700
600
500
400
300
200
100
0
0
5
10
15
20
25
no. waypoints

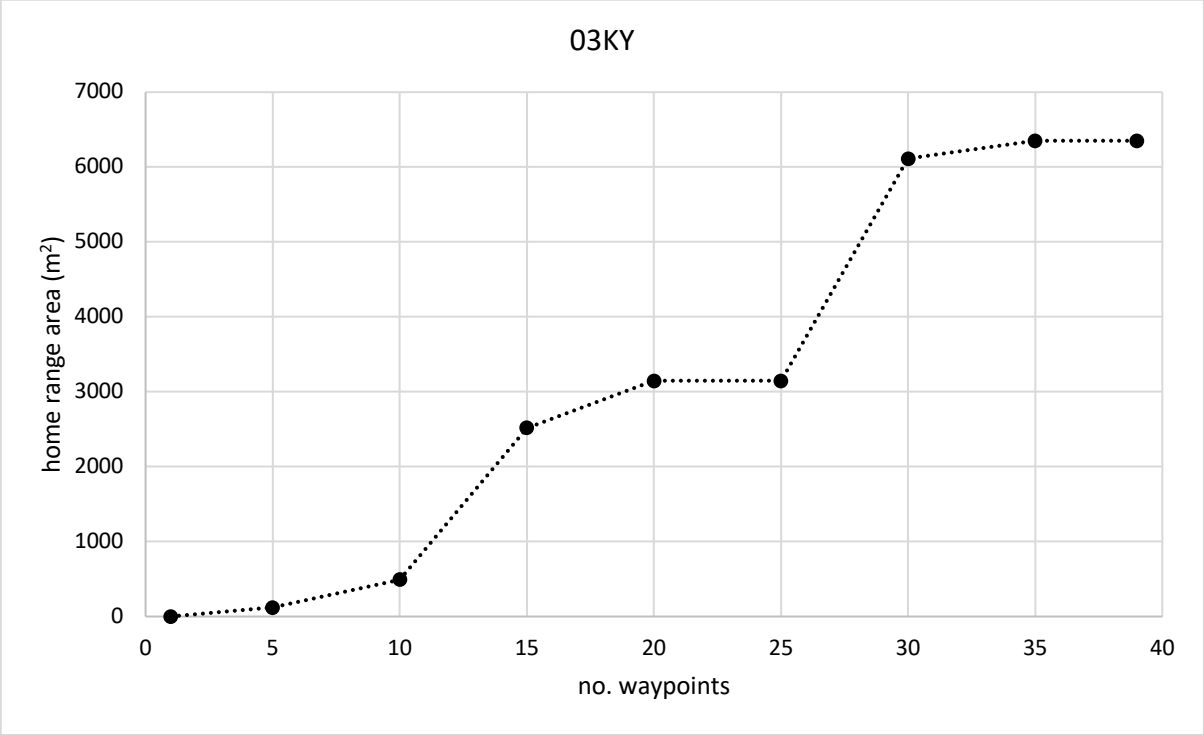

01FA
home range area (m2)
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
0
5
10
15
20
25
30
35
40
no. waypoints

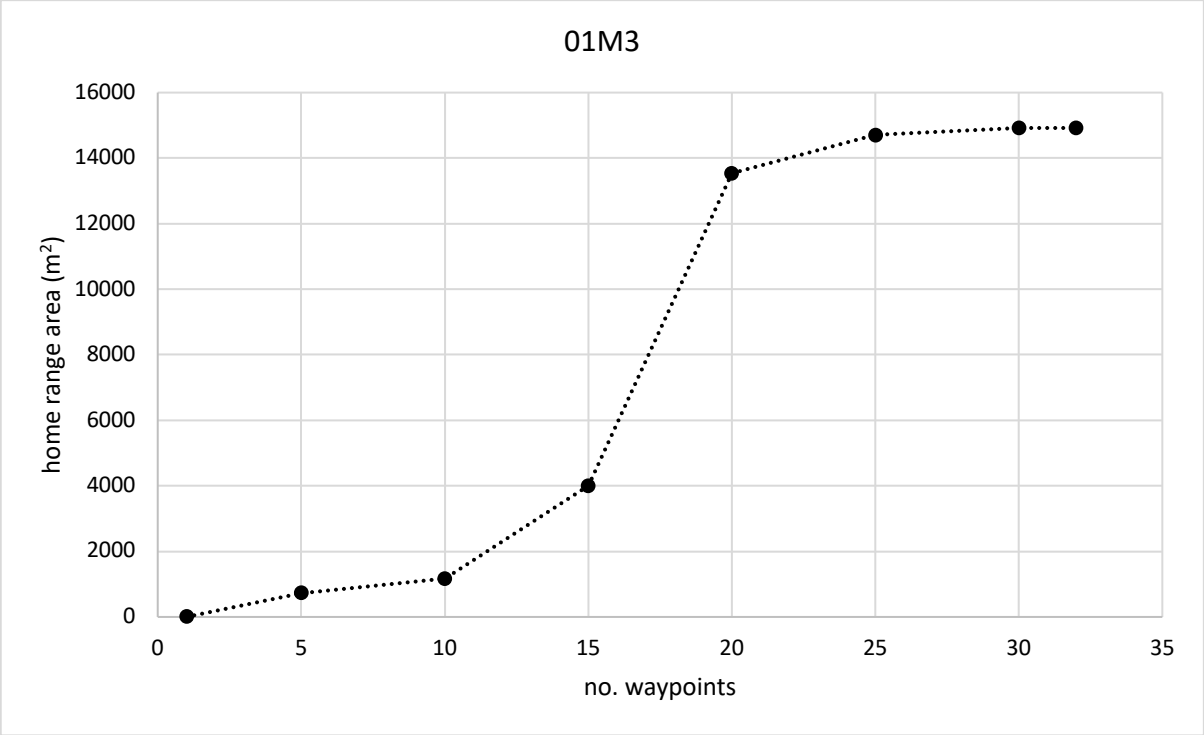

01H3
home range area (m²)
6000
5000
4000
3000
2000
1000
0
0
5
10
15
20
25
30
35
no. waypoints

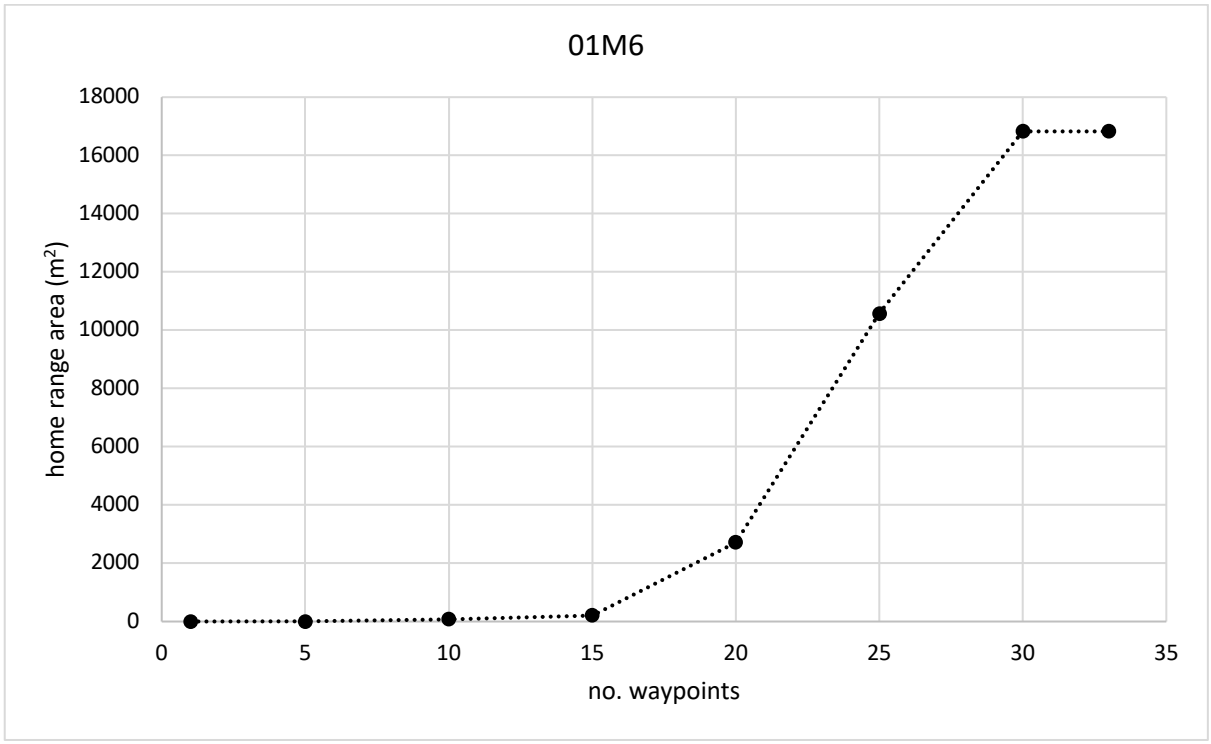

02H3
home range area (m²)
8000
7000
6000
5000
4000
3000
2000
1000
0
0
5
10
15
20
25
30
35
40
45
no. waypoints

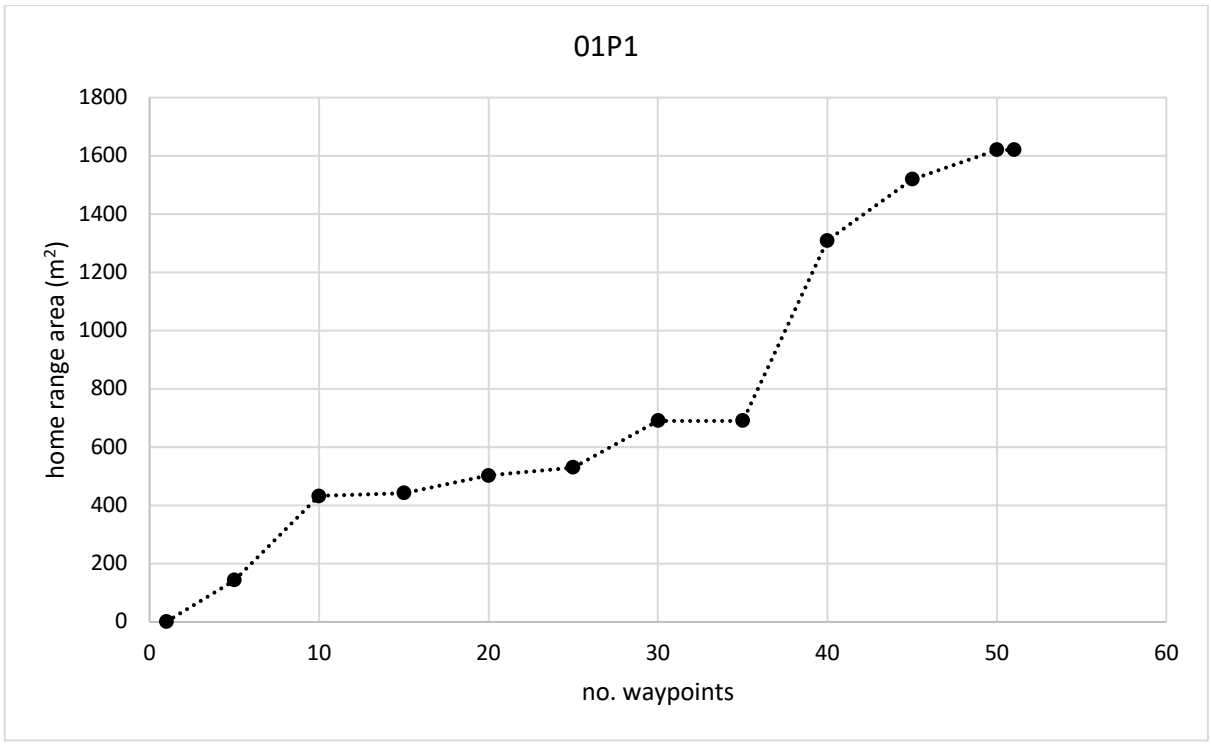

01KY
home range area (m²)
14000
12000
10000
8000
6000
4000
2000
0
0
5
10
15
20
25
30
35
40
45
50
no. waypoints

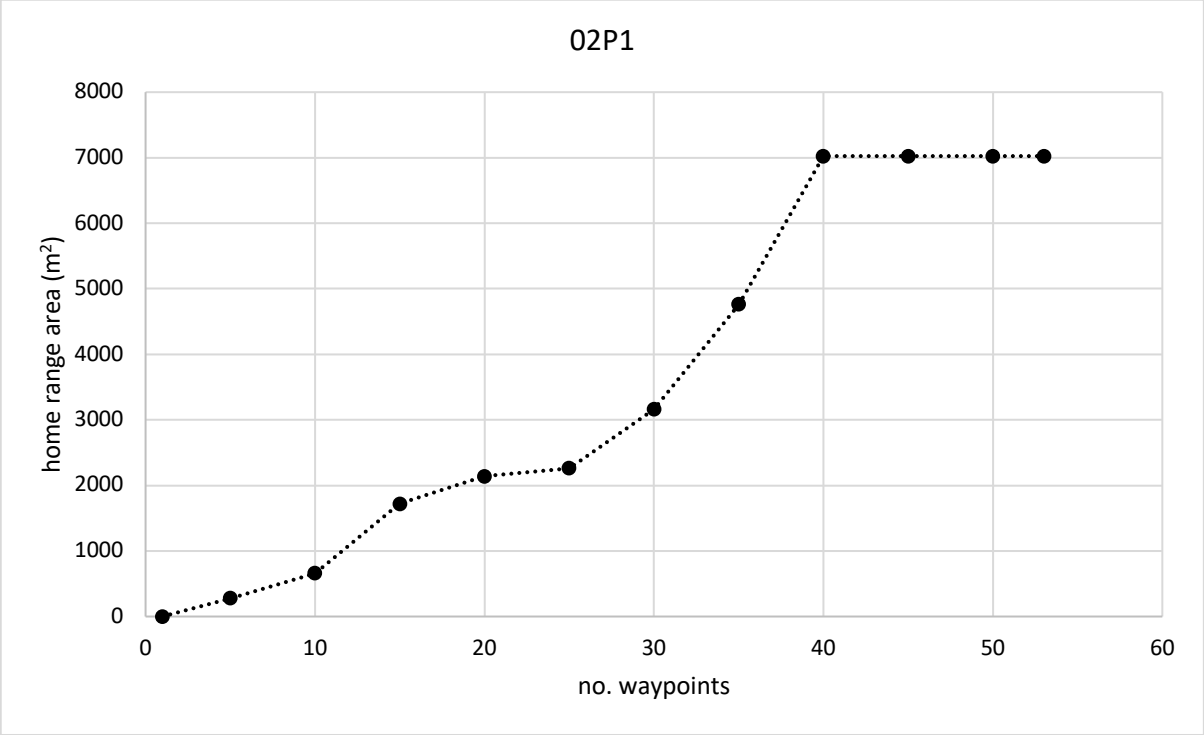

02KY
home range area (m²)
12000
10000
8000
6000
4000
2000
0
0
10
20
30
40
50
60
no. waypoints

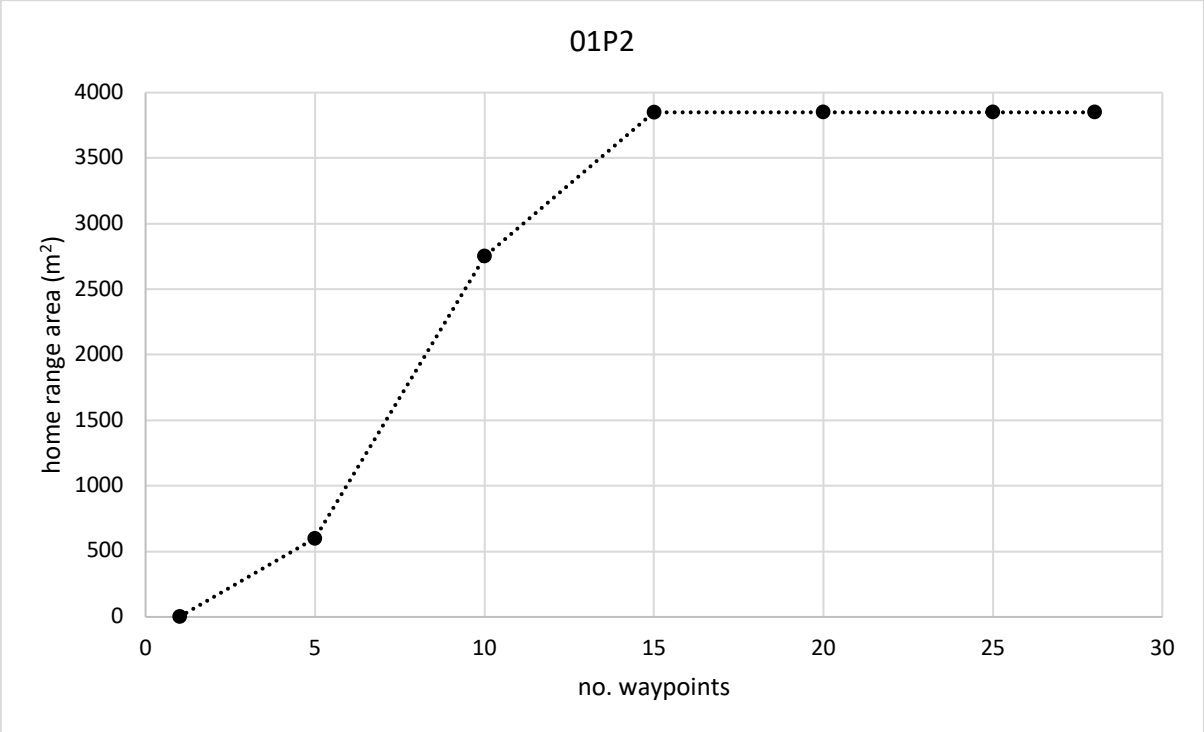

03KY
home range area (m²)
7000
6000
5000
4000
3000
2000
1000
0
0
5
10
15
20
25
30
35
40
no. waypoints

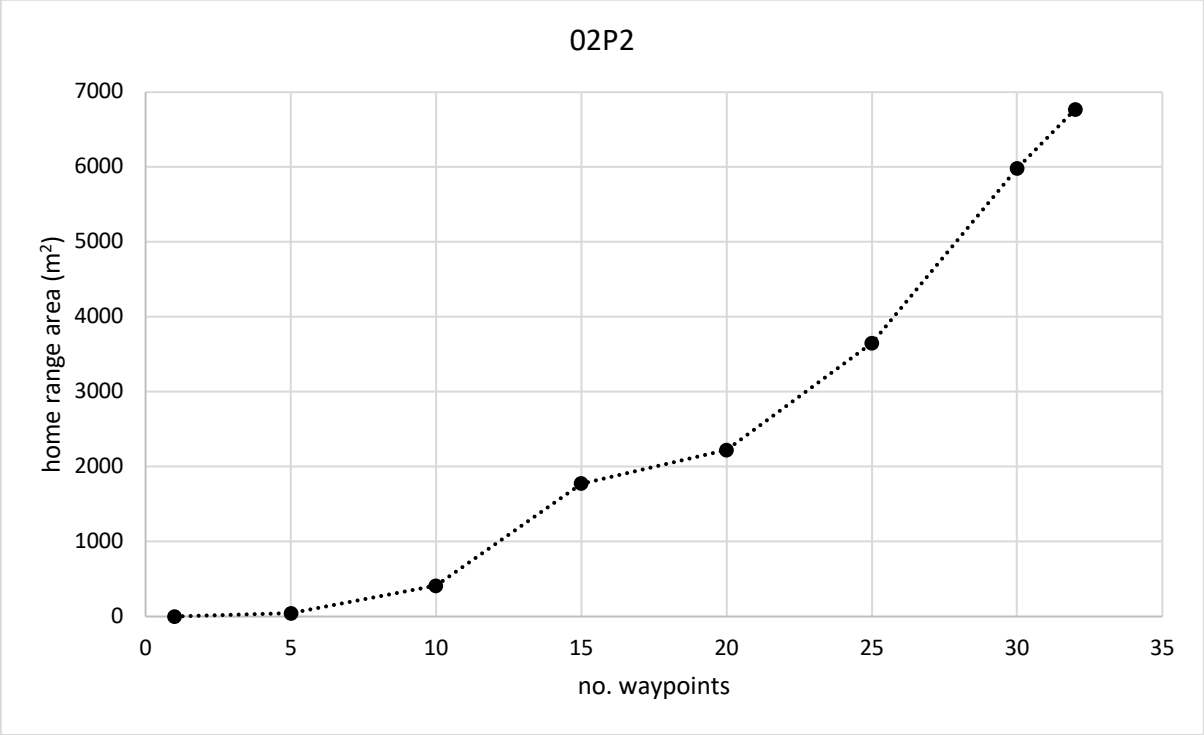

01M3
home range area (m²)
16000
14000
12000
10000
8000
6000
4000
2000
0
0
5
10
15
20
25
30
35
no. waypoints

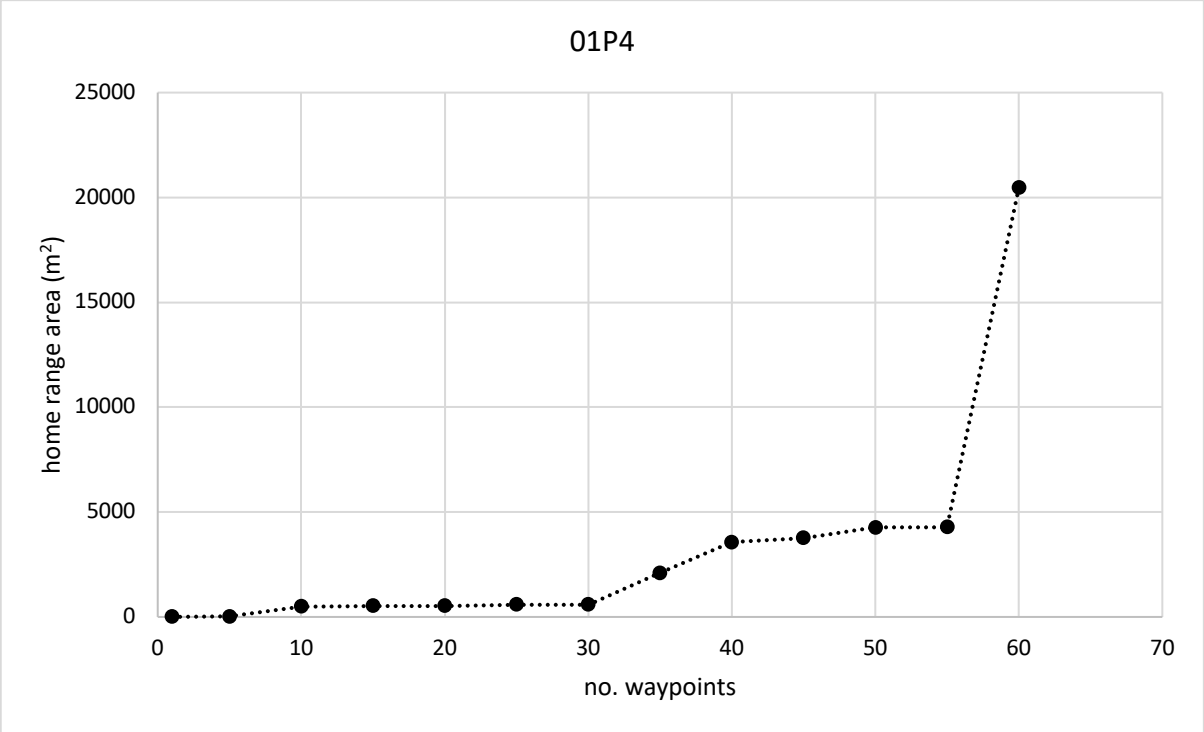

01M6
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
home range area (m²)
0
5
10
15
20
25
30
35
no. waypoints

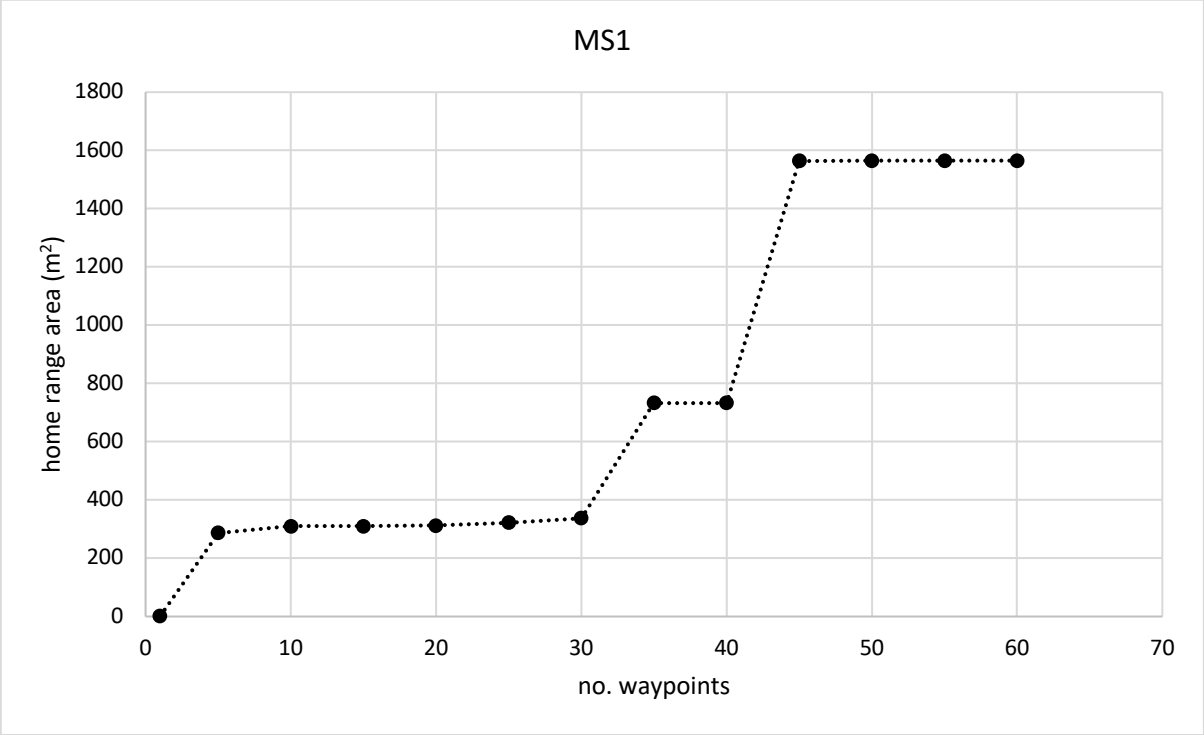

01P1
1800
1600
1400
1200
1000
800
600
400
200
0
home range area (m²)
0
10
20
30
40
50
60
no. waypoints

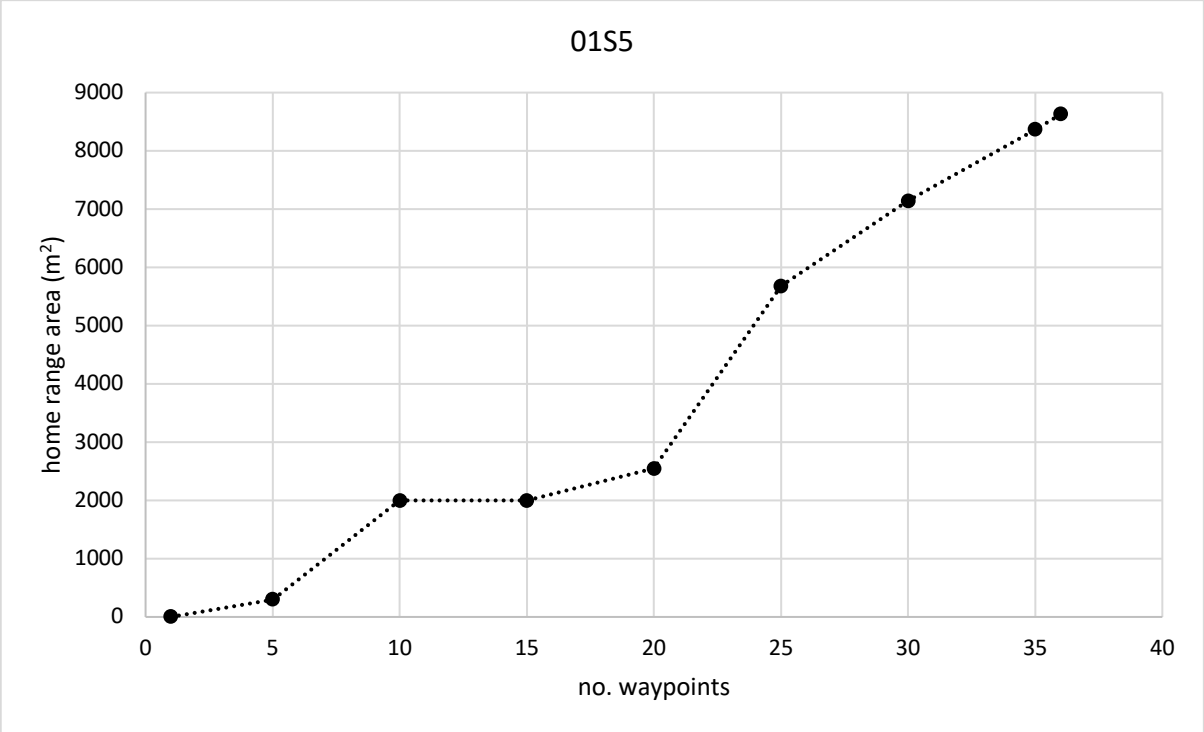

02P1
home range area (m²)
8000
7000
6000
5000
4000
3000
2000
1000
0
0
10
20
30
40
50
60
no. waypoints

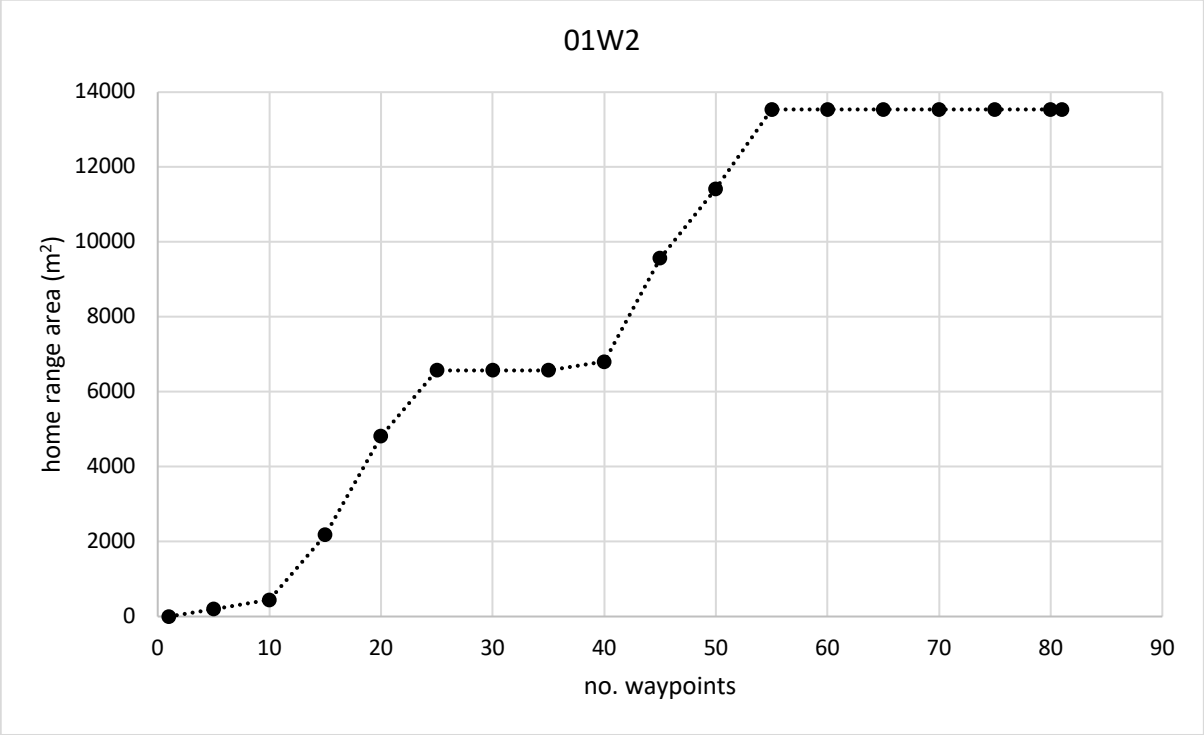

01P2
home range area (m²)
4000
3500
3000
2500
2000
1500
1000
500
0
0
5
10
15
20
25
30
no. waypoints

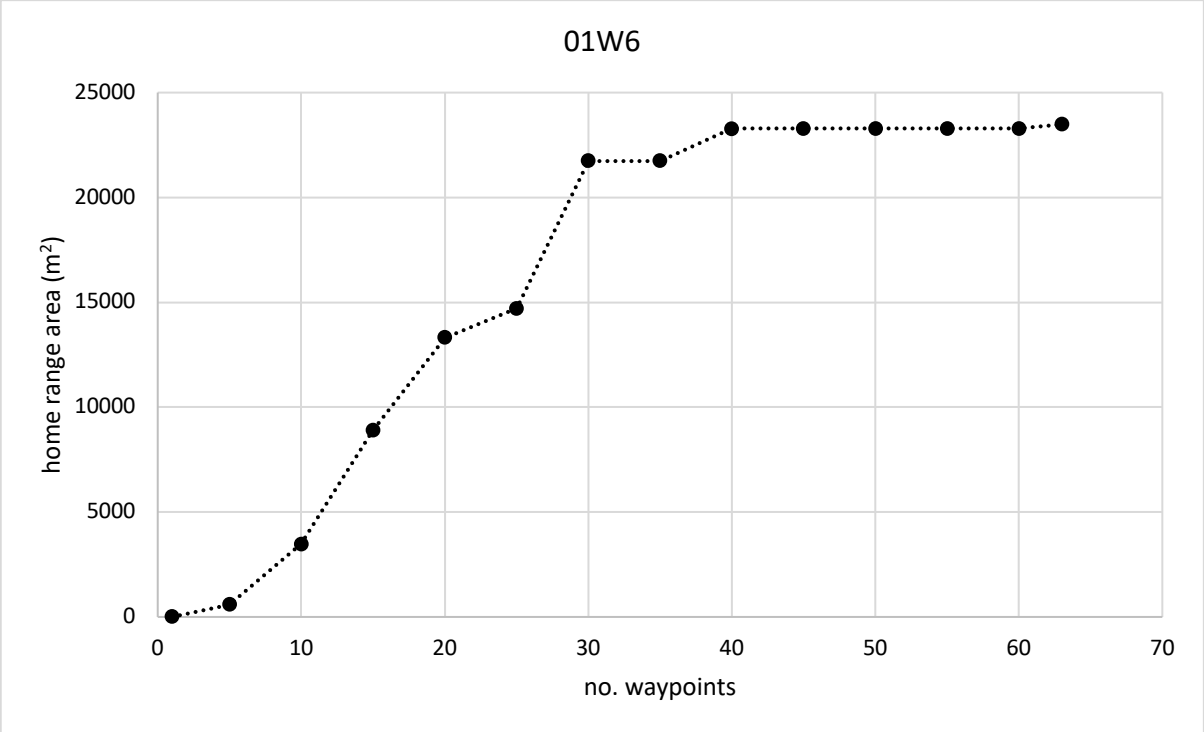

02P2
home range area (m²)
7000
6000
5000
4000
3000
2000
1000
0
0
5
10
15
20
25
30
35
no. waypoints


01P4
home range area (m²)
25000
20000
15000
10000
5000
0
0
10
20
30
40
50
60
70
no. waypoints


MS1
home range area (m²)
1800
1600
1400
1200
1000
800
600
400
200
0
0
10
20
30
40
50
60
70
no. waypoints


01S5
home range area (m²)
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
0
5
10
15
20
25
30
35
40
no. waypoints


01W2
14000
12000
10000
8000
6000
4000
2000
0
home range area (m²)
0
10
20
30
40
50
60
70
80
90
no. waypoints


01W6
25000
20000
15000
10000
5000
0
home range area (m²)
0
10
20
30
40
50
60
70
no. waypoints

GLOSSARY

bioregion	a landscape classification developed for Australia's National Reserve System, based on common climate, geology, landform, native vegetation and species information. There are 89 bioregions in Australia.
box-gum	certain *Eucalyptus* species, typically characterised by rough and/or tessellated bark known as box-type bark
critical weight range	mammals weighing between 35 and 5500 g, which are considered the most vulnerable to extinction
daily nest survival (DSR)	the daily probability of a nest surviving to the end of the study
derived native grassland	grassland associated with grassy woodland ecological communities
ecological trap	a habitat patch that disproportionately attracts birds, but is associated with poor survival or breeding success
fledge	to leave the nest as a juvenile bird
fledgling	a juvenile bird that has left the nest, but is still dependent on its parents for food and protection
hyperaggressive	a demeanour characterised by or tending toward unprovoked harassment or attacks
nestling	a juvenile bird in the nest
remnant	a patch of native vegetation that has not been cleared
restoration planting	a patch of native planted vegetation, usually in an agricultural landscape
site fidelity	the propensity of an animal to stay in, or return to, a habitat patch
travelling stock reserve	a Crown reserve, usually along a road, traditionally used to move stock between properties
windbreak	a linear restoration planting between two paddocks
woodland	an ecological community with an overstorey of trees up to 18 m high, and canopy cover of 20-80%